4.4.3 Discussion of PW-13 Diesel Contamination

The potential for microorganisms to degrade hydrocarbon (natural attenuation) contamination (diesel) in Well PW-13 and the two new wells, TRA-1933 and TRA-1934, is evaluated by examining the water chemistry. Diesel fuel is typically composed of hydrocarbons that have a range of 5 to 20 carbon atoms (C_5 to C_{20} hydrocarbons) with the majority of the diesel product in the C_9 through C_{20} range. In general, the C_5 through C_8 hydrocarbons are more soluble in water and mobile in the environment than the C_9 through C_{20} hydrocarbons. The C_5 through C_8 hydrocarbons include the BTEX components.

Dissolved oxygen is an important parameter for evaluating natural attenuation of hydrocarbons. Although dissolved oxygen data were not collected in March 2004 for PW-13, TRA-1933, and TRA-1934, dissolved oxygen was measured in October 2004. The dissolved oxygen readings for PW-13, TRA-1933, and TRA-1934 in October 2004 were 0.17, 0.21, and 0.70 mg/L, respectively. The low dissolved oxygen readings in PW-13, TRA-1933, and TRA-1934 in comparison to other perched wells (average 5.5 mg/L) suggest that aerobic respiration has depleted the amount of dissolved oxygen. If the background or natural dissolved oxygen concentrations for PW-13, TRA-1933, and TRA-1934 are similar to the average for the other perched wells at TRA, then the perched water has a significant aerobic assimilative capacity or natural attenuation capacity.

The biologically mediated processes involved in the degradation of hydrocarbons typically produce a shift from oxidizing to anaerobic (reducing) conditions in the groundwater. The shift from oxidizing to anaerobic conditions produces changes in groundwater chemistry that are reflected in the concentrations of redox-dependant species. The primary processes involved in the degradation of dissolved hydrocarbons are (1) aerobic respiration (dissolved oxygen concentration decrease), (2) anaerobic-denitrification (nitrate concentration decreases), (3) anaerobic-iron (III) reduction iron (+3) (dissolved iron concentration increases), (4) anaerobic-sulfate reduction (sulfate concentration decreases), and (5) anaerobic-methanogenesis (methane concentration increases) (EPA 1994, 1998). The degradation processes and changes in groundwater chemistry are shown on Figure 36. In addition, the creation of anaerobic conditions can lead to significantly increased concentrations of manganese, barium, and arsenic along with the production of methane. The significance of anaerobic processes can be evaluated by looking at concentrations of dissolved oxygen, nitrate, dissolved iron, and sulfate.

The elevated manganese concentration in Wells PW-13, TRA-1933, and TRA-1934 indicate that dissolved oxygen is depleted and therefore utilized to degrade hydrocarbons. Nitrate is typically utilized after dissolved oxygen is depleted. Nitrate concentrations in PW-13 are lower than the possible water source (SRPA water from TRA-03), but concentrations in the source are also low (less than 1 mg/L). Because nitrate is lower in concentration in PW-13 than TRA-03, it appears that nitrate is being at least partially utilized. In TRA-1933 and TRA-1934, nitrate is below detection limits and appears to have been utilized, suggesting that anaerobic biodegradation processes are active. However, without a good background value for nitrate, it is not possible to fully evaluate the utilization of nitrate in PW-13, TRA-1933, or TRA-1934. Because manganese concentrations are elevated and iron concentrations are only slightly elevated in all three wells, this would indicate that the redox conditions are not favorable for the reduction of sulfate. This would imply that sulfate is not being utilized to a large extent by bacteria and that the sulfate concentration in PW-13 is only marginally affected. The sulfate concentrations in TRA-1933 and TRA-1934 are lower than in PW-13, suggesting that sulfate is utilized to a slightly greater extent in these wells than in PW-13. Barium concentrations are significantly elevated in PW-13, but not in TRA-1933 or TRA-1934. Arsenic concentrations are within the natural range for all three wells.

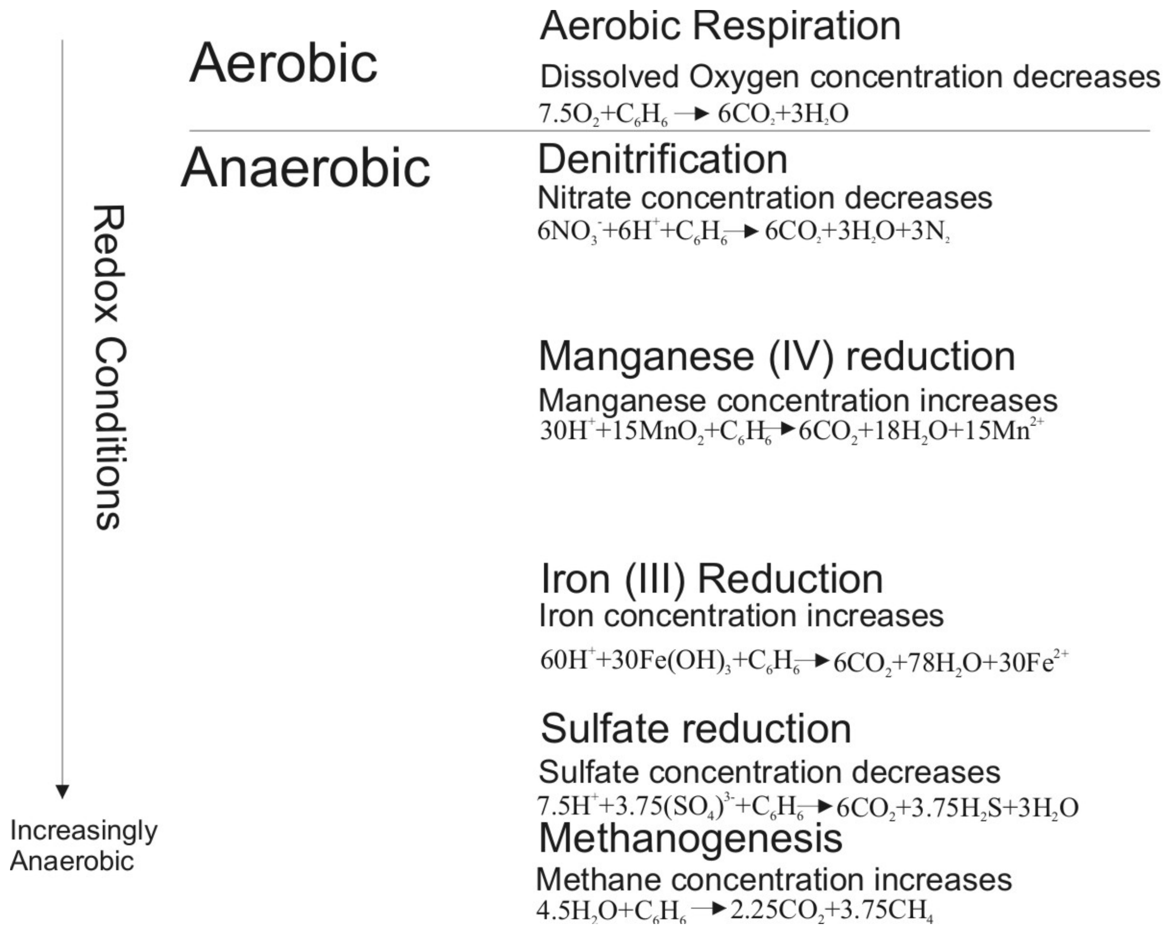

Figure 36. Schematic illustrating the natural attenuation processes involved in the degradation of hydrocarbons. (Note that the spacing between processes reflects the relative redox change and possible reactions [from (EPA 1998)] for benzene [C_6H_6] are given as examples.)

Anaerobic degradation may not have fully developed in Wells PW-13, TRA-1933, and TRA-1934—as evidenced by extensive sulfate depletion, high dissolved iron concentrations, and the presence of methane—because the aerobic assimilative capacity of the perched water is sufficient to prevent this zone from developing. If floating petroleum product is from a release in the early 1980s, then most of the mobile and soluble hydrocarbon fraction of the product would have had time to have partitioned from the residual product. The current low gasoline-range organics and relatively low diesel-range organic concentrations in the perched water indicate that the more mobile and easily degradable hydrocarbons have probably been removed from the residual floating product. In addition, the mobile and soluble hydrocarbons are the most biodegradable components of diesel. Thus, the remaining residual product in perched water is probably composed of low-solubility hydrocarbons that are resistant to natural attenuation processes.

The BTEX and gasoline-range organics data indicate that natural attenuation processes have removed the C_5 through C_8 hydrocarbons that are the more soluble and mobile hydrocarbons. Although no samples of the residual floating petroleum product have been analyzed, the product is probably composed mostly of greater than C_9 hydrocarbons that are not readily degradable. If recovery of the remaining petroleum fraction (floating product) is required, physically pumping (either actively or passively) may be the best alternative. Free-phase hydrocarbons in nonaqueous-phase liquid (floating product) are not readily degradable by microorganisms and are only slightly affected by natural attenuation processes. The highly variable occurrence of floating product in PW-13 would favor passive recovery of the petroleum product.

Alternatively, the hydrocarbons could have migrated away from the floating product, but several studies have demonstrated that dissolved hydrocarbon plumes are susceptible to biodegradation processes and the data collected from this investigation suggest that as well (EPA 1994). The biodegradation of dissolved hydrocarbons, principally BTEX, is mainly limited by the supply of electron acceptors (dissolved oxygen, nitrate, iron (III), sulfate, and carbon dioxide). The authors of the *Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water* (EPA 1998) concluded that in most, if not all, hydrogeologic environments there appears to be an adequate supply of electron acceptors.

4.4.4 Evaluation of Flux from the Vadose Zone into the Aquifer

The extent of the CWP's influence on the SRPA can be qualitatively evaluated by looking at a map of the sulfate plume in the SRPA (Figure 37). The distribution of sulfate in the SRPA in 1991 points to the CWP as the primary source of the sulfate plume, because the highest sulfate concentrations are downgradient (see Figure 8) of the CWP (Figure 37). The sulfate plume could also in part originate from the former Chemical Waste Pond, since sulfate was elevated in the MTR test well in 1991, which is located near the former Chemical Waste Pond. However, it is uncertain if the high sulfate and cation concentrations in the MTR test well are from the CWP or from the former Chemical Waste Pond. The influence of the CWP and former Chemical Waste Pond can be seen on a plot of sulfate concentrations in the perched water in 1991 (Figure 38). The small increase in sulfate above background (Well TRA-03 is used for background based on the hydraulic gradient) in USGS-058 suggests that most of the water from the CWP impacts the SRPA south-southwest of USGS-58.

Two aquifer wells, USGS-065 and TRA-07, downgradient of the CWP show higher $\delta^{18}O$ and δD values than the other aquifer wells. The higher $\delta^{18}O$ and δD values along with elevated sulfate concentrations indicate influence from the CWP. The higher $\delta^{18}O$ and δD values probably reflect mixing of seasonal $\delta^{18}O$ and δD values at depth in the vadose zone. This would imply that the average travel time from the perched zone to the aquifer is greater than 6 months to allow for mixing over several seasons (at least spring to fall). However, not enough data exist to fully evaluate this travel-time estimate. The USGS data for USGS-065 collected on May 16, 1991, indicated similar $\delta^{18}O$ and δD values (-16.90 and -133.0 per mil), as determined in this investigation (USGS 1999). Well TRA-08 plots in between the δD and $\delta^{18}O$ range for the SRPA and the values for the CWP (Figure 33). This suggests that the water in Well TRA-08 probably reflects mixing of water from the CWP with water from the SRPA.

Elevated tritium and chromium concentrations also are associated with elevated sulfate concentrations in the TRA-07, USGS-065, and TRA-08 SRPA wells. This could indicate that while the CWP is not the source of these contaminants, water from the CWP is aiding the transport of these contaminants to the aquifer. In addition, USGS-076 shows sulfate and chromium above background in 2004 and has had tritium detections in the past, but tritium was not sampled for in USGS-076 in 2004.

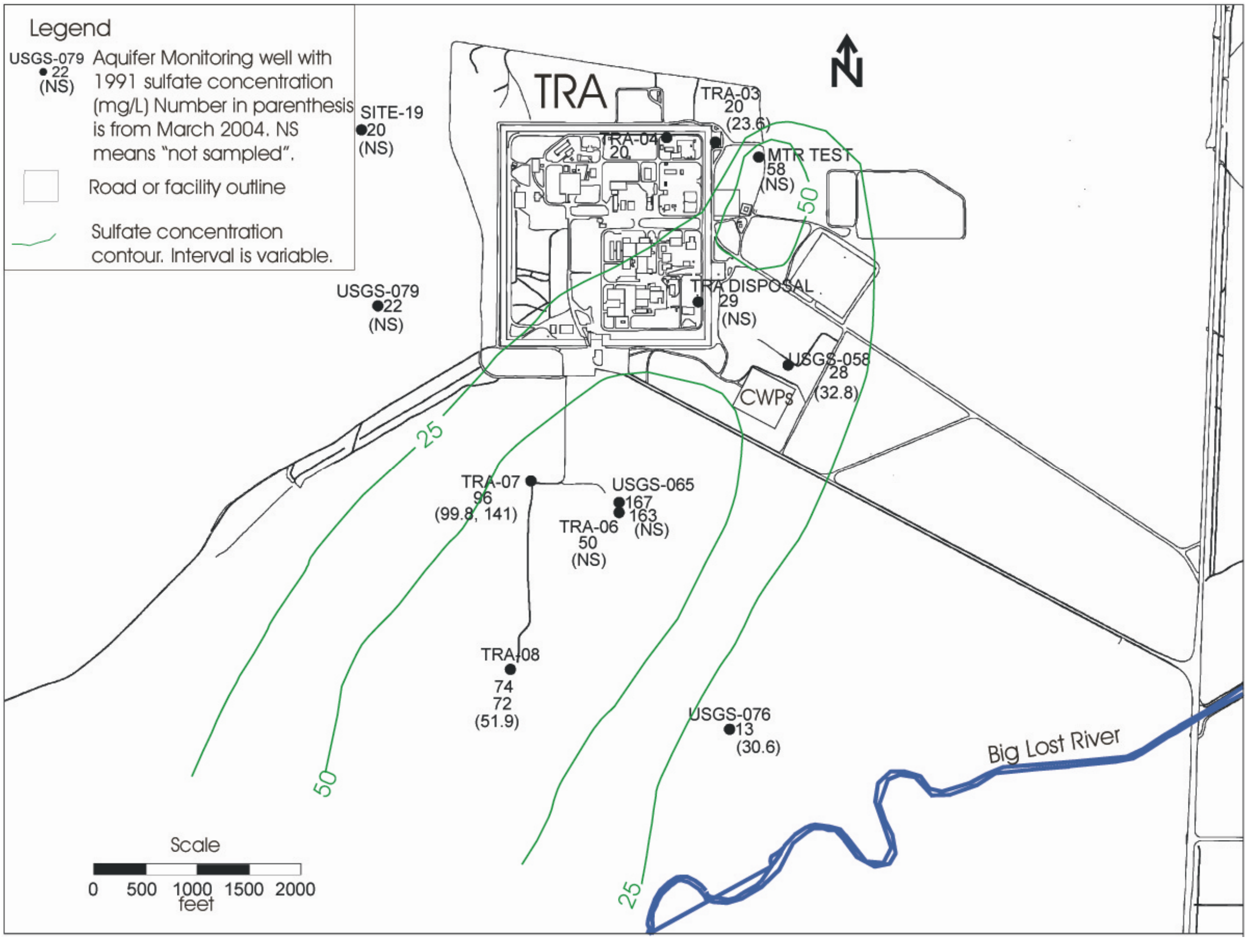

Figure 37. Distribution of sulfate in the Snake River Plain Aquifer in 1991 and 2004.

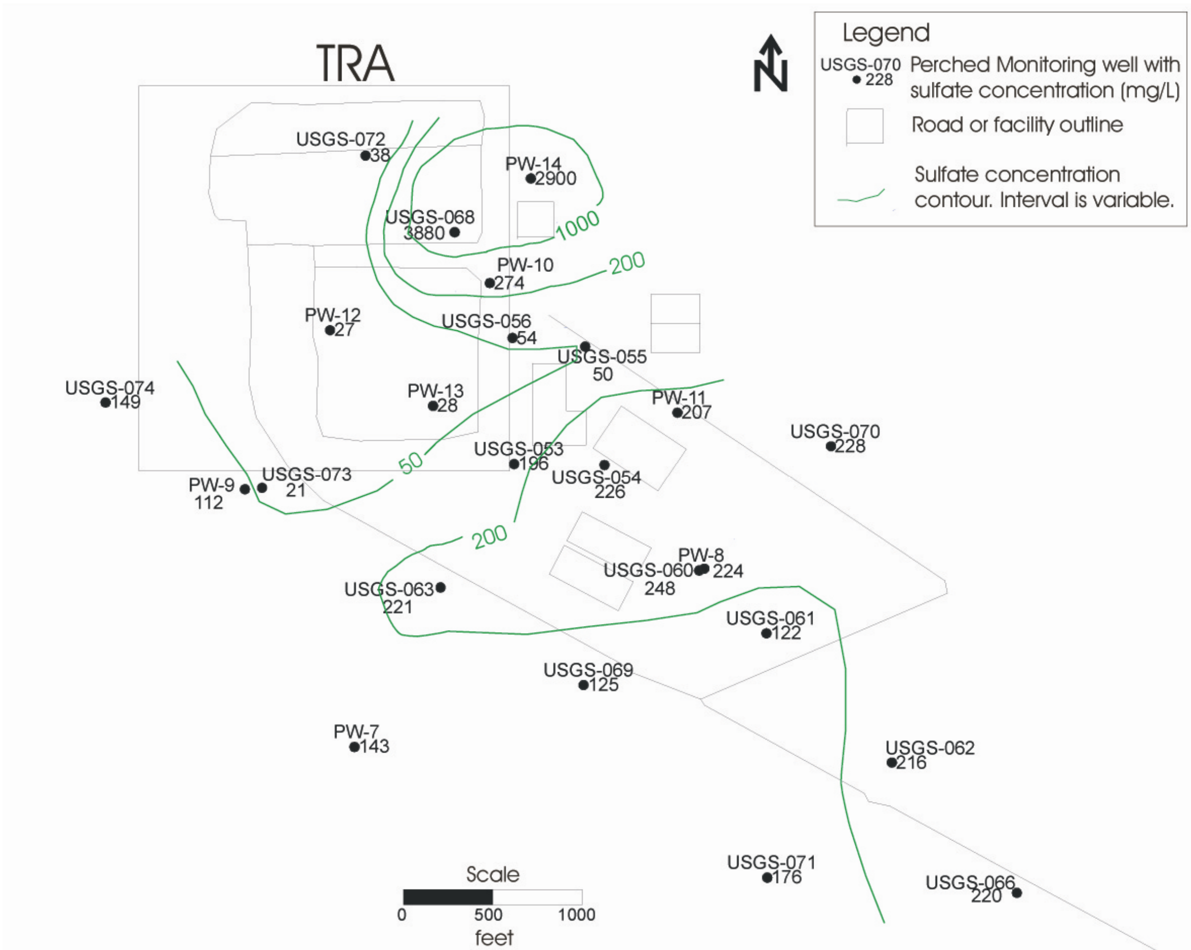

Figure 38. Distribution of sulfate in the perched water in 1991.

4.5 Summary and Conclusions

The contaminant, water-quality, and stable-isotope data indicate that the deep perched water beneath TRA consists of several types of water from multiple sources rather than a single, homogenous perched-water body. These sources include the obvious source of water from the CWP. In addition, water-chemistry data from Wells PW-12, PW-13, and USGS-072 indicate a source derived from leakage from underground (raw water) piping with some precipitation input. The perched water near these wells appears to have a separate geochemical signature from water derived from the CWP. The contaminant, geochemical, and hydraulic head data for PW-9, USGS-073, PW-12, and PW-13 suggest that the deep perched zone might actually consist of upper and lower perched-water zones rather than a single deep perched-water body.

Wells USGS-068 and PW-14 reflect contamination from the former Chemical Waste Pond with a local water source (precipitation and raw water lines). Well PW-14 has dried up since the Chemical Waste Pond was taken out of service, indicating that the primary water source for this area has been eliminated.

The source of Co-60 (see Section 5) appears to be a carbon-steel warm waste line. Transport of the Co-60 contamination observed in PW-12 might be caused by a leaking raw water pipeline aided by precipitation that carries the Co-60 from the soil around the warm waste line.

The water chemistry data indicate that natural attenuation processes have removed the less than C_8 components (the more soluble fraction) of the diesel product near Wells PW-13, TRA-1933, and TRA-1934 with the possible result that the floating product is composed of the less soluble, greater than C_8 hydrocarbons that are more resistant to natural degradation processes.

Although the CWP is not the source of contaminants, water infiltrating from the CWP could aid the migration of contaminants. This is indicated by the fact that the wells in the SRPA that have the highest contaminant concentrations (tritium and chromium) also show the strongest influence from the CWP.

4.6 Recommendations

The possibility of the deep perched water consisting of multiple perched-water bodies needs to be examined in terms of the site geology. The site geology should be evaluated with respect to the presence of perching layers and the influence of stratigraphy on perched water.

A sample of the floating product should be collected from Well PW-13 (if it reappears) and analyzed to verify that the less than C_8 fraction of the product is gone and that it is composed primarily of greater than C_8 hydrocarbons. This would also confirm that an additional source is not contributing to the floating product.

5. COBALT-60 IN WELL PW-12

5.1 Introduction

The First Five-Year Review Report (DOE-ID 2003) identified the unexplained increase of Co-60 in the PW-12 perched-water well as an issue. Cobalt-60 is a gamma-emitting isotope with a half-life of 5.2 years (EG&G Idaho 1991a). The PW-12 perched-water well is located near the center of the TRA facility, just west of the TRA-661 building. In March 2003, samples collected from PW-12 indicated that the Co-60 activities in the well had exceeded the MCL of 200 pCi/L. It should be noted that with regard to Co-60, the only well in question is PW-12. Cobalt-60 was not detected in samples collected from the aquifer during the period of evaluation for the five-year review (1990 to 2003) or during sampling events following the review. The investigation into the increase in activities at PW-12 indicates that the increase is most likely the result of remobilization of residual contamination in the subsurface by changing hydrogeologic conditions causing higher levels of contamination to move toward the well. Process knowledge indicates that the most recent use of a waste line in the vicinity of PW-12 was in 1996, coincident with the closure of the Hot Cells located in TRA-632. Recent fluctuations in Co-60 concentrations in Well PW-12 might be attributed to contaminant mobilization from three nearby CERCLA sites with known Co-60 contamination. The investigation concluded that the selected remedies listed in the OU 2-13 ROD (DOE-ID 1997a) remain protective of human health and the environment.

This section addresses the issue of Co-60 contamination in the PW-12 perched-water well. The history of PW-12 and Co-60 measurements is provided in Section 5.2. Potential sources for that contamination are discussed in Section 5.3. Conclusions are provided in Section 5.4 and recommendations are presented in Section 5.5.

5.2 History of PW-12

Well PW-12 was completed during the 1990 investigation of the perched water beneath TRA. The well was cored to a total depth of 141.5 ft bls. The well was completed with a screened interval from 108 to 128 ft bls using a 4-in. stainless-steel, wire-wrapped screen. A filter pack of 10 × 20 sand was installed from 102.5 to 131.8 ft bls. The associated well casing is 4-in. stainless steel. A bentonite seal extends from the top of the sand to 97.5 ft bls with cement grout from 97.5 ft bls to land surface. A 10-in. protective surface casing was installed during the grouting and extends from approximately 3 ft above ground surface to 39.6 ft bls. The alluvial/basalt interface is located at 39 ft bls. A 4-ft-thick zone of red sand and cinders was encountered at 138 ft bls. The remainder of the drilled interval consists of basalt.

As part of the Waste Area Group 2 groundwater monitoring plan, Well PW-12 has been sampled quarterly to semiannually since 1993 for gamma emitters, including Co-60. The history of Co-60 activities in water samples collected from PW-12 is displayed in Figure 39. Measured Co-60 activities show a decreasing trend from 1995 to late 2002; Co-60 activities were below detectable limits during some sampling events. Measured Co-60 activities began to increase in October 2002, culminating in an activity of 330 pCi/L measured during the spring of 2003. After peaking in March 2003, subsequent samples collected from Well PW-12 indicated decreasing activities. By October 2003, the Co-60 activities were at 62.2 pCi/L and the March 2004 sample was 52.7 pCi/L, continuing the decreasing trend.

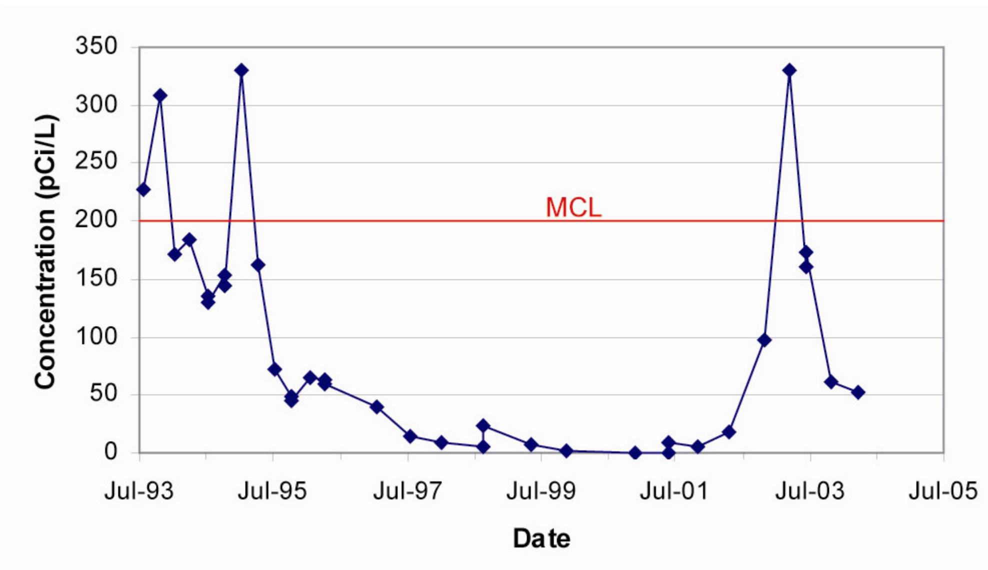

Figure 39. Activities of cobalt-60 in Well PW-12.

5.3 Proximal Cobalt-60 Contamination

Buildings and pipelines in the vicinity of Well PW-12 are known to have historically contained various radionuclides, including Co-60. This section discusses numerous buildings, pipelines, and CERCLA sites near PW-12 that could be sources for the Co-60 contamination observed at PW-12. The data are insufficient to identify the specific source, but the recent decline in Co-60 activity measured in water from PW-12 suggests that the Co-60 is from one of the historical releases described in this section. It should be noted that Co-60 contamination has not been detected in the aquifer during the evaluation period of the five-year review (1990–2003) or following the review. In addition, Well PW-12 was the only well identified during the five-year review period with a potential issue pertaining to Co-60. Although Co-60 has been detected in other perched-water wells, Well PW-12 is the only well to display an increase in the reported activity levels. The rest of Section 5.3 describes the history of CERCLA sites with known Co-60 contamination proximal to PW-12 and other potential sources for Co-60 contamination at TRA.

5.3.1 Brass Cap Area

The Brass Cap Area (see Figure 40) is identified as a CERCLA site under the OU 2-13 ROD (DOE-ID 1997a) and lists Co-60 as a COC. The Brass Cap Area is located in the center of TRA, near the TRA-630 building and southeast of Site TRA-19 between TRA-635 and TRA-632. The location of the brass cap is approximately 250 ft southeast of Well PW-12. The original source of contamination at the Brass Cap Area was identified as the 3" HDA-630 hot waste line. The release of contaminants was identified in 1985, but it was believed to be the result of damage to the pipeline that occurred during the installation of a new firewater line in 1978 (EG&G Idaho 1985a). The contaminants were later remobilized away from the site in the summer of 1985 when a pressurized firewater line separated (EG&G Idaho 1985b). A brass marker was placed in the concrete to designate the area of subsurface contamination. The contamination under the concrete was determined to extend to approximately 10 ft bls. After the contamination was discovered, the leaking waste line was repaired and contaminated soil in the immediate proximity of the repaired waste line was removed. The excavation was backfilled with clean soil and resurfaced with concrete (DOE-ID 2003).

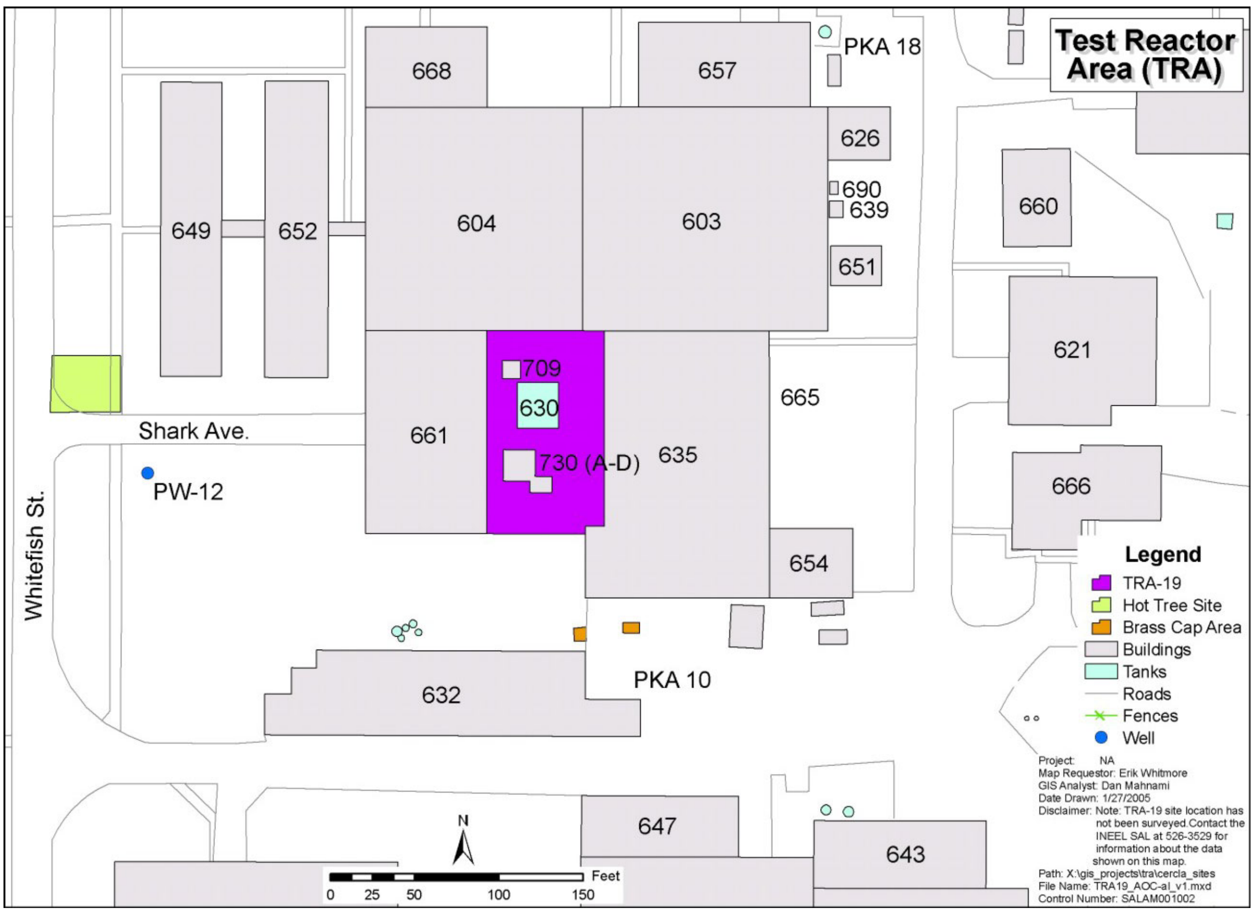

Figure 40. Map of Comprehensive Environmental Response, Compensation, and Liability Act sites in the area of PW-12.

5.3.2 Hot Tree Site

The TRA-43 CERCLA site, Hot Tree Site (Figure 40), is located in the center of TRA, west of the TRA-649 building and approximately 50 ft northwest of Well PW-12. Surface screening of the spruce tree's branches by RCTs indicated that it had been contaminated with gamma-emitting radionuclides. The tree was removed, boxed, and disposed of in May 1994. After removal of the tree, soil borings were collected for field screening approximately 2 ft bls in the area of the tree. In addition, the root system was surveyed. Three additional surface soil samples were collected and submitted to the Radiation Measurements Laboratory for analysis. The results of sampling were evaluated in the OU 2-13 baseline risk assessment. The risk assessment showed that an unacceptable risk does not exist at this site because of low contaminant concentrations in the soil. The site is restricted to industrial usage, because the calculated current residential excess cancer risk is greater than 1E-04 for 30 years because of potential exposure to Cs-137. The highest radiologically contaminated area was west of the tree, indicating that an abandoned warm waste line, 4" WDC-641, was the source of contamination. The warm waste line is a Duriron line that originated from the Gamma Facilities Building (TRA-641), approximately 10 ft west and 6 ft bls from the Hot Tree Site. Waste transferred through the line consisted of low-pressure, demineralized water. Since it was believed that the line was drained and there was no leak test performed, this line was submitted for evaluation and acceptance under the *Federal Facility Agreement and Consent Order for the Idaho National Engineering Laboratory* (DOE-ID 1991) as new CERCLA Site ID# TRA-61 on March 9, 2000 (DOE-ID 2003). It was recommended that this site did not meet the criteria for acceptance as a new site. The responsible project managers concurred with this recommendation between March 21 and April 2, 2001.

Data regarding the Hot Tree Site, listed in the OU 2-13 ROD (DOE-ID 1997a), indicate that although Co-60 is not a COC, Co-60 contamination was detected in samples collected at the site. The Hot Tree Site is located approximately 50 ft northwest of Well PW-12.

5.3.3 Soil Surrounding Tanks at TRA-630 (TRA-19)

The *Comprehensive Remedial Investigation/Feasibility Study for the Test Reactor Area Operable Unit 2-13 at the Idaho National Engineering and Environmental Laboratory* (DOE-ID 1997b) identified Co-60 as one of the possible COCs for the soil surrounding the tanks at TRA-630 (TRA-19) (Figure 40). The site, TRA-19, located near the Catch Tanks, is approximately 220 ft northeast of Well PW-12. According to the OU 2-13 ROD (DOE-ID 1997a), contamination of the soil surrounding the tanks was the result of a leak from a radiologically contaminated waste drain line (WDC-641) originating within the Gamma Facility Building (TRA-641) or a possible release from the Catch Tanks. The Catch Tanks were replaced in 1985–1986 and were found to be intact, and no leaks are suspected from either the new tanks or the old tanks. However, recent investigations have determined that there may have been releases from multiple sources in the courtyard.

The 4" HDC-604B warm waste line ran from the radiochemistry laboratories in the TRA-604 building to the TRA-630 Catch Tanks. The HDC-604B line also was used to transfer waste from TRA-661 to TRA-630. The HDC-604B line was removed from service in 1985. The HDC-604A line is documented as having leaked in Facility Change Form 8.9.2-4 (EG&G Idaho 1991b). The line was repaired after leaking in 1985 and was in service until it was cut and abandoned in 1991. The chemical composition and volume of the release were not recorded and are unknown. The HDC-604B line is located approximately 200 ft northeast of PW-12 at its nearest point.

A 4-in. Duriron line, 4" HDC-632 (located approximately 200 ft east of PW-12) was used to transfer hot waste from Building TRA-632 to TRA-630. Building TRA-632, also known as the Hot Cells, was used for irradiation experiments and processes, utilizing various materials, including Co-60. The line was used from the 1950s until 1996. There is no evidence to indicate that the 4” HDC-632 line has leaked. Investigations conducted by the Voluntary Consent Order (VCO) Program, including excavations near the pipeline, have not located any contamination outside the pipeline. The 2” HDA-661 pipeline is a 2-in. stainless-steel hot waste transfer line. At its nearest point, the line is approximately 175 ft due east of PW-12. The line was installed in 1959 or 1960 and was used until 1996. The line transferred hot waste from TRA-661 to the TRA-713 hot waste storage tanks. A release from this line was discovered in 1986. During excavation for the TRA-730 tank vault, the line was found to be leaking (Briscoe 1986). The line was repaired after the new vault was installed. A total of 1,024 ft^3 of contaminated soil was removed from the excavation. The soil was removed to 10 ft below grade. A Geiger Mueller detector indicated that contamination extended an additional 3 ft beneath the excavation.

5.3.4 Additional Known Cobalt-60 Contamination

Several other sites with known Co-60 contamination have been documented at TRA (DOE-ID 1997b). Although it is unlikely that these sources are currently contributing to the increase in the Co-60 activity noted in Well PW-12, they should still be identified as potential contributors because of the recorded presence of Co-60. Sites listed below are more distal from the source relative to those listed in the previous subsections. The CERCLA sites with recorded Co-60 contamination are listed below:

- TRA-03B (Warm Waste Pond sediments)
- TRA-04 (Warm Waste Retention Basin)
- TRA-08 (Cold Waste Pond)
- TRA-13 (Sewage Leach Pond and Soil Contamination Area)
- TRA-15 (Soil Surrounding Hot Waste Tanks at TRA-613).

5.4 Conclusions

Because of the recent increase and subsequent decrease in the Co-60 activities in PW-12, coupled with similar trends in historical activities, monitoring of this well will be continued to ensure that the remedy remains effective. Despite the recent increase and decrease in the activities at this location, the remedy still seems to be effective at this time. The decreasing activities reported in samples collected from PW-12, coupled with the short half-life of Co-60, suggest that the unexpected increase is not related to a new source from an ongoing release. The brief increase likely is due to the mobilization of a pulse of Co-60 from residual contamination to PW-12 or due to hydrogeologic mechanisms discussed in Section 6 of this document.

Mobilization may occur when recharge pathways to perched water change in response to variations in precipitation and irrigation or potentially from leaks in raw water lines within the area. Although the water lines are not currently documented as leaking, an analysis of geochemical data from samples from perched-water wells in March 2004 may indicate leakage from pipes containing raw water (see Section 4). Water samples collected from several wells, including PW-12, displayed chemical signatures closely resembling that of a sample collected from the TRA-03 supply well, an aquifer well located near the northeast corner of TRA. Further analysis also showed that the chemical signature of the water in these wells is consistently different than in wells near to the CWP and that this signature has been present since at least 1991. Currently, there are raw, potable, demineralized, and firewater lines near PW-12 and the three referenced CERCLA sites. If residual Co-60 exists in the geologic matrix (alluvium or basalt) around or near one or more of these water lines, and the water line is leaking, it is possible for the contamination to be mobilized. Analytical data seem to indicate that the source water is most likely raw water, ruling out demineralized water lines. Likewise, changes in the location and rate of infiltration from precipitation or irrigation might carry a pulse of Co-60 to PW-12 as the water moves through residual contamination. Alternatively, the increase in the activities of Co-60 in PW-12 could be the result of hydrogeologic mechanisms discussed in Section 6 of this report. Section 6 discusses the hydrogeologic mechanisms that may influence fluctuations of certain analytes in the perched water at TRA.

It should also be noted that the March 2003 sample result of 330 pCi/L does not seem to be anomalous or erroneous. The sample data received Level “A” validation and was not flagged by the laboratory or during validation. When viewed in relation to past and subsequent results for PW-12, the March 2003 data point does not seem atypical for this well. At least one other such spike in Co-60 activity has been observed in this well. A similar increase and decrease in the activities of Co-60 occurred in 1995 (Figure 39).

5.5 Recommendations

Continued monitoring in accordance with the Groundwater Monitoring Plan (DOE-ID 2004a) is recommended. The increase of Co-60 noted in the five-year review and the subsequent decrease seems to be in line with past monitoring data. However, continued monitoring is warranted in accordance with the accepted groundwater monitoring plan, but no additional activities are recommended at this time. The pipelines believed to be responsible for the three CERCLA sites near PW-12 are currently under investigation by the VCO (TRA-630 Area Catch Tanks Closure Project) Program (DOE-ID 1997b); however, it should be recognized that VCO is not performing their investigation as a response to the Co-60 levels. The VCO investigation is related to independent actions relating to the removal of piping and building material at these locations. Piping within these sites is subject to Resource Conservation and Recovery Act (42 USC § 6901 et seq.) closure under VCO-5.8.d, TRA-004, and TRA-11. Soil samples within existing CERCLA sites will be conducted under Resource Conservation and Recovery Act closure, and data collected will be used during CERCLA activities to determine if the contingent excavation remedy specified in the ROD is required. Consequently, the most plausible sources for Co-60 are currently under investigation. It is further recommended that that the VCO activities and findings for the aforementioned investigations be tracked by CERCLA personnel so that the new information can be used to assist in the interpretation of groundwater monitoring results. Although the VCO is conducting an independent investigation for a distinct purpose, their investigation may provide information useful in assessing groundwater contamination at TRA.

6. POTENTIAL MECHANISMS AFFECTING PERCHED CONTAMINANT CONCENTRATIONS

The First Five-Year Review Report (DOE-ID 2003) identified the occurrence of steady or increasing activities of Sr-90 in the PW-12, USGS-054, USGS-055, and USGS-070 perched-water wells as an issue. It was further recommended that several mechanisms be evaluated as being potential causes for these unexplained trends. Those mechanisms include (1) adsorption/desorption occurring with changing perched-water levels, (2) changing flow pathways in response to lining of the Warm Waste Pond and/or fluctuations in discharge to the CWP (or between alternating cells), (3) seasonal variations of natural infiltration at a local scale, (4) variations in recharge from unidentified manmade sources, (5) lateral flux from the Big Lost River, or (6) new leaks of contamination from unidentified sources. This section considers possible mechanisms that might explain the unexpected trends in Sr-90 measurements. The issue of unpredicted concentration trends is introduced in Section 6.2. The potential explanatory mechanisms identified in the First Five-Year Review Report are discussed in Section 6.3. Recent research findings suggest another possible mechanism that is introduced in Section 6.4. Conclusions are provided in Section 6.5.

6.1 Unpredicted Concentration Trends

Contaminant-concentration data from the TRA perched-water monitoring network were evaluated as part of the five-year review for the period 1997–2002. Measured chromium, tritium, Sr-90, and Co-60 concentrations decreased during this period at most of the deep perched-water wells. However, contaminant concentrations did not decrease in several of the wells. Contaminant concentrations were expected to decrease in response to changing disposal practices and natural processes, including radioactive decay and water/rock interactions. Strontium-90 concentrations in water from Wells PW-12, USGS-54, USGS-55, and USGS-70 (see Figure 1) increased or remained about the same during the five-year period (Figure 41). As discussed further in Section 5 of this report, the Co-60 concentration in water from Well PW-12 increased (Figure 39) in 2002. Additionally, chromium concentrations in water from several TRA perched-water wells rose during the early 1990s, coincident with perched-water-level increases (Figures 42 and 43).

6.2 Potential Mechanisms Identified for the First Five-Year Review

Mechanisms that could result in unanticipated contaminant-concentration trends include (1) adsorption/desorption occurring with changing perched-water levels, (2) changing flow pathways in response to remediation and fluctuations in discharge to the CWP, (3) seasonal variations of natural infiltration at a local scale, (4) variations in water recharge from unidentified manmade sources, (5) lateral flux from the Big Lost River, or (6) potential leaks of water from unidentified sources. The unanticipated contaminant-concentration trends also could result from a combination of these mechanisms. The remainder of this section discusses these potential mechanisms.

6.2.1 Adsorption/Desorption of Contaminants of Concern

Concentrations of the contaminants in groundwater may fluctuate as aqueous geochemical equilibria cause adsorption or desorption of the contaminants onto and off of the geologic media. This section discusses adsorption/desorption relationships and their potential effect on perched water beneath TRA. Mobility of a specific contaminant in perched groundwater at TRA is dependent on the chemical valence of the contaminant, geochemical processes within the subsurface, and fluctuations in perched-water levels.

6.2.1.1 Mobility of Selected Contaminants of Concern at the Test Reactor Area. The mobility of chromium is dependent on its valence state. Under natural conditions, chromium can occur in two valence states: Cr(III) and Cr(VI). The Cr(III) exists in the environment under reducing conditions and is highly immobile. Hexavalent chromium (Cr[VI]) exists under oxidizing conditions in the environment as chromate anion (CrO_4^{2-}), which is very mobile (Oliver et al. 2003). The Cr(VI) is estimated to account for 89% of the total chromium in groundwater beneath TRA (EG&G Idaho 1989). Because a large fraction of the total chromium in the subsurface beneath TRA occurs in the hexavalent state, chromium is one of the most mobile of the COCs at TRA, second only to tritium (Dames and Moore 1992a).

Both Co-60 and Sr-90 exist in solution as divalent cations. As a result, both are relatively immobile and retarded during transport (Dames and Moore 1992a).

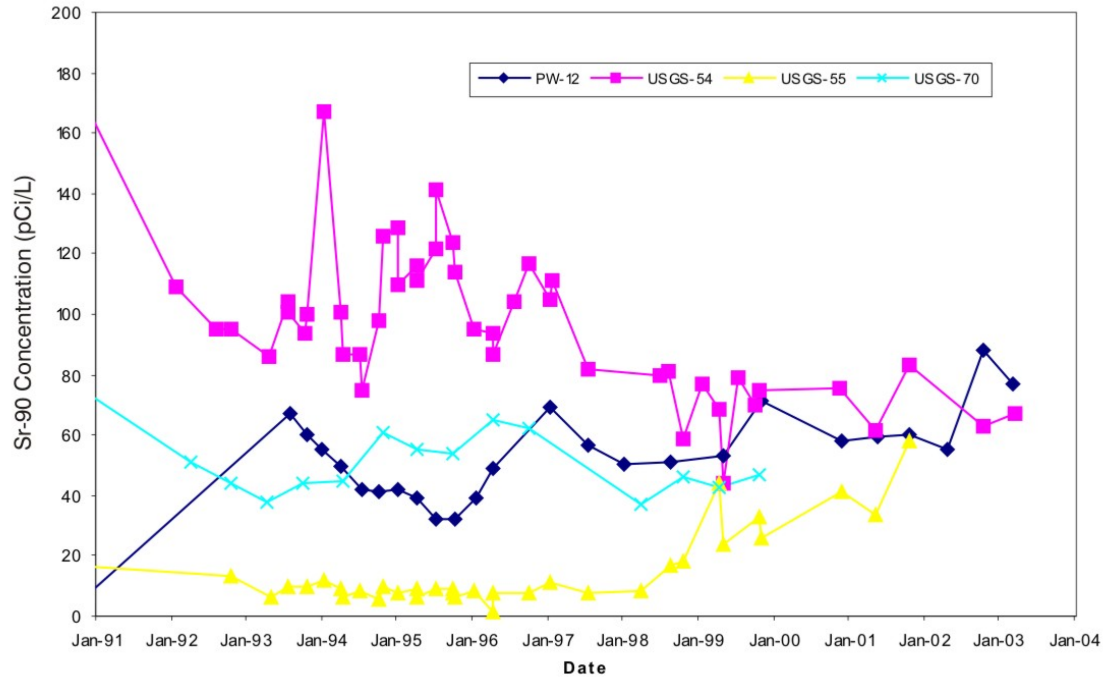

Figure 41. Strontium-90 concentrations in deep perched-water wells proximal to the Warm Waste Pond.

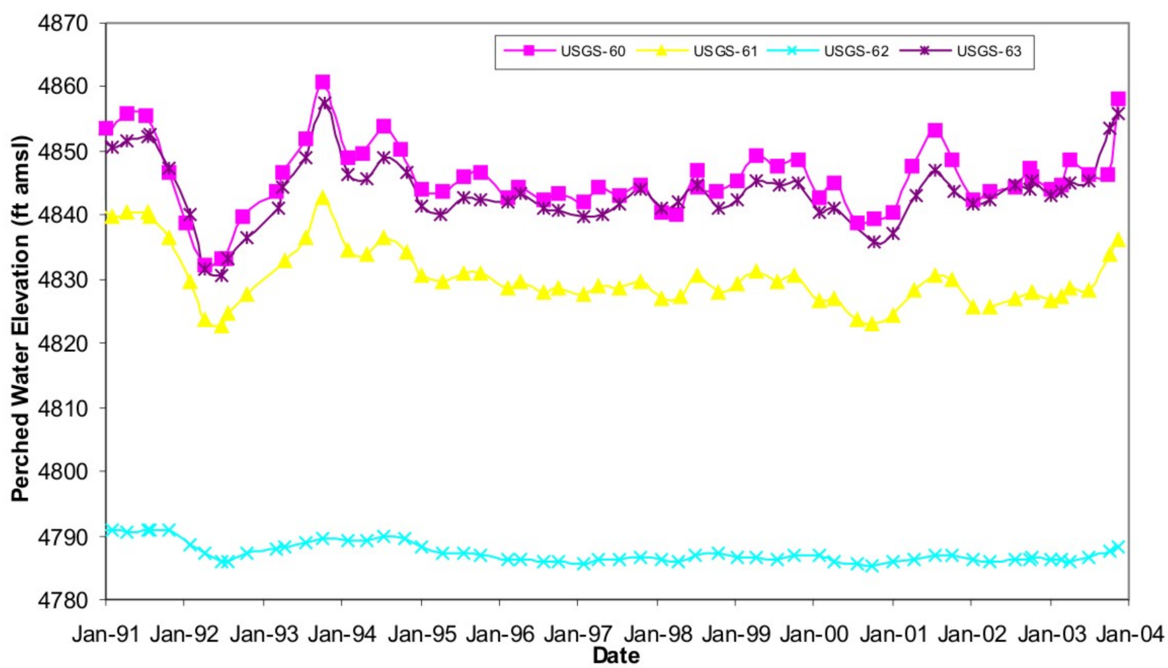

Figure 42. Hydrographs for the USGS-60, USGS-61, USGS-62, and USGS-63 perched-water wells.

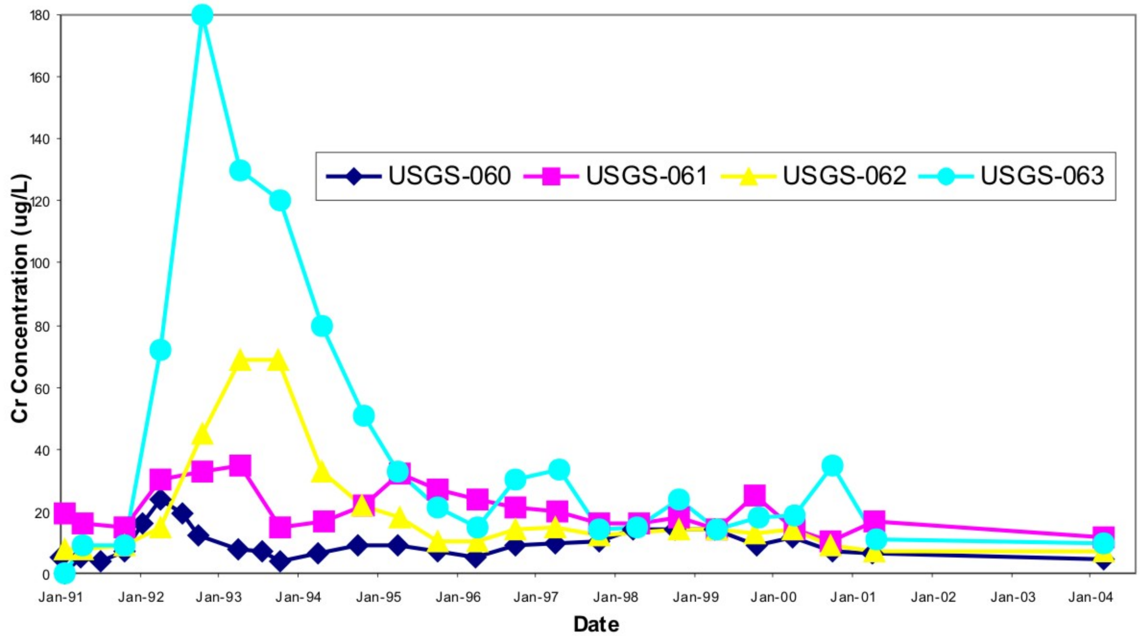

Figure 43. Chromium trending in Wells USGS-60, USGS-61, USGS-62, and USGS-63.

6.2.1.2 Geochemical Processes that Control Adsorption/Desorption. Adsorption occurs because of the presence of electrically charged sites on solid surfaces that attract oppositely charged dissolved species. The sorptive capacity of a solid is primarily controlled by the density of charged surface sites and the surface area of particles on which those surface sites occur. A sorbent phase with a net negative surface charge is said to have a cation exchange capacity (i.e., dissolved species with a positive charge are adsorbed), whereas a sorbent phase with a net positive surface charge is said to have an anion exchange capacity (i.e., anions are sorbed). Surface charge may be permanent and independent of a solution's composition or can vary with changes in solution composition. Surface charge is permanent in many clay minerals, while surface charges of metal oxyhydroxide minerals are highly dependent on the pH of the solution; they are net positive at low pH and net negative at higher pH (Langmuir 1997).

The pH inflection point at which neither cations nor anions are preferentially adsorbed is material dependent and is known as the point of zero net proton charge (PZNPC). Published PZNPCs for Na-feldspar fall near the pH of natural waters (5.2<PZNPC<6.8) (Langmuir 1997). The PZNPCs of several iron oxides and oxyhydroxides (including goethite and hematite) also fall near the pH of natural waters. Minor fractions of each of these minerals may occur in basalt at the INEEL. As a result, small fluctuations in the pH of soil and groundwater beneath TRA may lead to adsorption or desorption of a COC if the PZNPC is crossed.

In addition to the effects of pH on adsorption/desorption relationships, other components of solution chemistry could affect the degree to which sorbed species are desorbed. Adsorption of a dissolved ionic species is always part of an exchange reaction that involves a competing ionic species. Ion exchange models of adsorption are often employed to describe these as an equilibrium process (Langmuir 1997).

The ion exchange concept can be described in terms of exchange equilibrium between two similar ions, such as Ca^{2+} and Mg^{2+} (Langmuir 1997), as shown in Equation (2):

$$Ca^{2+} + MgX = Mg^{2+} + CaX \tag{2}$$

Where Mg^{2+} and Ca^{2+} are the dissolved ion concentrations and MgX and CaX are the fractions of each ion in the exchanger (solid) phase, the equilibrium or exchange constant for this reaction is shown in Equation (3) below:

$$K_{ex} = \frac{[Mg^{2+}][CaX]}{[Ca^{2+}][MgX]} \tag{3}$$

Equation (2) may be rearranged as follows in Equation (4):

$$[Mg^{2+}] = \frac{K_{ex}[Ca^{2+}][MgX]}{[CaX]} \tag{4}$$

Equation (3) demonstrates that the concentration of Mg^{2+} in solution is directly proportional to the concentration of Ca^{2+} in solution and the mass of Mg^{2+} sorbed to the solid phase. In other words, as the concentration of dissolved Ca^{2+} increases, the concentration of dissolved Mg^{2+} also increases as Ca^{2+} replaces Mg^{2+} on the solid phase and drives Mg^{2+} from the solid phase into solution.

In the context of perched water beneath TRA, increases in the dissolved concentration of cations such as Ca, Mg, Na, or heavy metals might be expected to drive sorbed cations such as Co-60 or Sr-90 from exchange sites and into solution. The result would be an increase in the concentration of Co-60 or Sr-90 in solution. Similar exchange-site competition can occur between anions such as chromate, sulfate, bicarbonate, and nitrate if a net positive surface charge existed on the solid phase.

6.2.1.3 Effect of Perched-Water-Level Fluctuations on Sorption. Increases in perched-water levels may be a mechanism by which water of one chemical composition (i.e., the perched water) is mixed with water of another (e.g., the soil water in the unsaturated material that becomes inundated as perched-water levels rise). In the case of chromate, if the perched water contains higher levels of competing anions than the soil water, chromate may be released from the previously unsaturated material by ion exchange according to an equilibrium reaction similar to that presented in Equations (2) through (4).

In addition, if the pH of the perched water is lower than the PZNPC and the pH of the soil water immediately above the perched-water zone is higher than the PZNPC, an anion species might desorb from the previously unsaturated solid material and go into solution.

It is difficult to quantify the effects of this mechanism without extensive study on the sorptive capacity of minerals contained in the geologic materials beneath TRA and equilibrium relationships for the species competing for sorption and desorption. Much of this information may be available for clay minerals and nonradioactive dissolved species; however, the effects of basalt mineralogy and the presence of radionuclides on adsorption/desorption relationships are sufficiently unique that a large amount of original research would have to be conducted to quantify this mechanism.

Differences in water chemistry that may cause adsorption (or desorption) are difficult to define, because only samples of the perched-water body are currently collected. Water contained in the vadose zone immediately above the perched-water zone is not sampled and its chemistry is not known.

6.2.2 Changing Flow Paths in Response to Wastewater Disposal Changes

Variable discharge rates to TRA wastewater infiltration ponds might have impacted water chemistry in perched-water wells for several reasons. First, changing discharge rates to different infiltration ponds might have affected the wastewater flow pathways from the ponds to perched-water bodies. Second, the hydraulic character of the perched-water body itself might have been modified by variable surface discharges. Each of these scenarios is expected to affect mass transport in the vadose zone. These scenarios are discussed in Sections 6.3.2.1 and 6.3.2.2, respectively.

6.2.2.1 The Effect of Wastewater Disposal on Flow Paths from the Land Surface to Perched-Water Bodies. Wastewater disposal changes to the TRA infiltration ponds might have affected flow paths between the ponds and perched-water bodies. These changes might have been reflected in COC concentration trends. Figures 42 and 43 contain water-level elevation and Cr concentration data from perched-water wells in the southeastern portion of TRA during 1990–2002. The Cr concentrations presented in Figure 43 were taken from both USGS records and TRA CERCLA-mandated sampling records. The data presented are an example of concentration variability in response to perched water-level changes. Water discharge to the CWP (Figure 44) can be compared to the concentration and water-level data shown in Figures 42 and 43. Low discharges to the CWP were observed between 1991 and 1993. The WWP was still being used for disposal at that time, although no notable trend in discharge volumes was observed. During this period, water levels declined in the USGS-60, USGS-61, USGS-62, and USGS-63 perched-water wells. Increased Cr concentrations were observed during a low-discharge period (1991–1993) in water samples collected from the same wells (Figure 43).

Based on these data, reduced discharge to the CWP might have acted to increase the concentration of selected COCs at certain wells despite the fact that none of the COCs were present in wastewater disposed of at the CWP. This response suggests an interplay between the flow-path set from the CWP and the flow-path set from the WWP. The COCs were disposed of at the WWP. Therefore, flow paths between the WWP and the deep perched zone should contain most of the residual COC contamination. As the contribution of flow along the CWP flow paths was decreased, concentrations of the COCs increased as the relative contribution from the WWP was increased (i.e., decreased dilution).

6.2.2.2 Perched-Water Flow Paths. Variable discharge to surface recharge sources may have locally resulted in changes to the physical character of the deep perched-water body itself. In the scenario presented in the previous section, changes in discharge to the CWP were shown to have effects on both perched-water levels and chromium concentrations at several wells. It is also plausible that additions of water from the surface to the deep perched-water body might result in local water-level increases. For example, localized mounds in the perched-water surface could develop beneath fractures and flow paths along which recharge occurs only during specific surface-discharge conditions. Similarly, discontinued flow through a flow path between the surface and the deep perched zone could result in a localized depression in the perched-water surface. These localized mounds and depressions would not be observable because of the coarse spacing of wells.

Localized perturbations to the surface of the perched-water body might change local hydraulic gradients within the perched-water body. These gradient changes might result in movement of water and contaminants from one part of the perched-water body to another. These possible fluxes could explain erratic contaminant concentrations with time in water from a monitoring well. However, these changes to the local hydraulic gradient in the perched-water zone would not increase or decrease the total mass of a particular contaminant in the entire perched-water body; rather, contaminant mass would simply be shifted from one location to another. Concentration trends influenced by this mechanism would be further affected by dispersion, sorption, and diffusion as the perched water is shifted between locations.

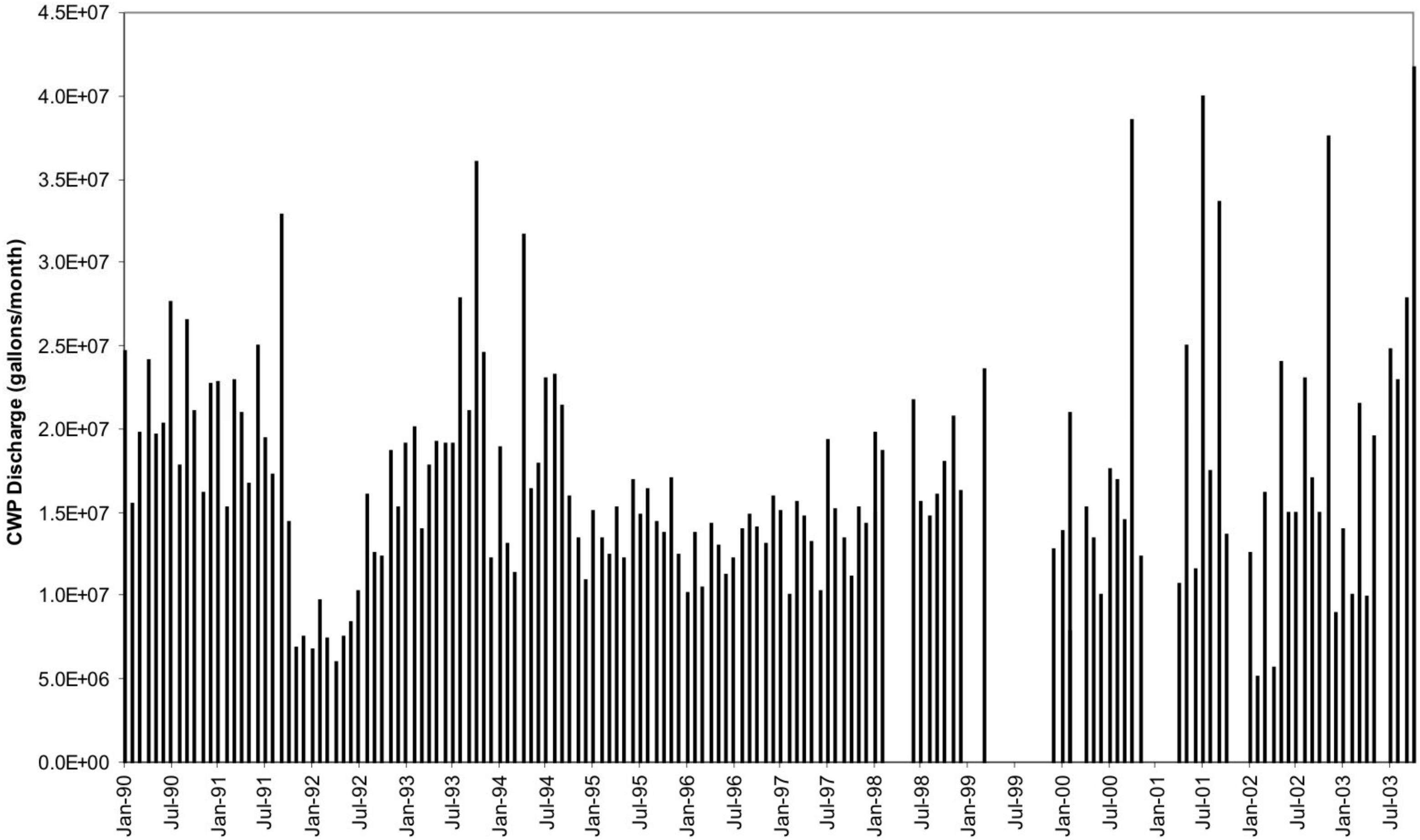

Figure 44. Monthly discharge to the Cold Waste Pond from 1990 through 2002.

6.2.3 Seasonal Variations of Natural Infiltration on a Local Scale

Seasonal variations in natural infiltration might result as run-off from spring snowmelt accumulates in surface depressions and ditches. If naturally derived water from a filled surface depression infiltrates into the perched groundwater body, this infiltration might modify flow paths and contribute to contaminant dilution. This mechanism is very similar to that presented in Section 6.3.2.2 with the exception that discharge variability will occur on a seasonal basis.

If this mechanism was an important factor in influencing concentration trends in the deep perched zone, notable concentration trends should have developed during the mid- to late-1990s. From 1996 through 2000, abnormally moist weather conditions persisted in the region. Figure 45 is a time series plot that contains cumulative precipitation and Sr-90 concentration data for four perched-water wells that have shown flat or increasing concentration trends in recent years. The three shaded intervals depicted in Figure 45 indicate notably moist periods. During the third shaded interval (April through June 1999), Sr-90 concentrations decreased in water collected from Wells USGS-54 and USGS-55. The lower Sr-90 concentration in those wells might be the result of dilution through localized infiltration of precipitation. Outside of that short-term fluctuation, no pronounced trends are apparent; however, the Sr-90 sampling frequency is generally insufficient to resolve seasonal trends resulting from the infiltration of precipitation.

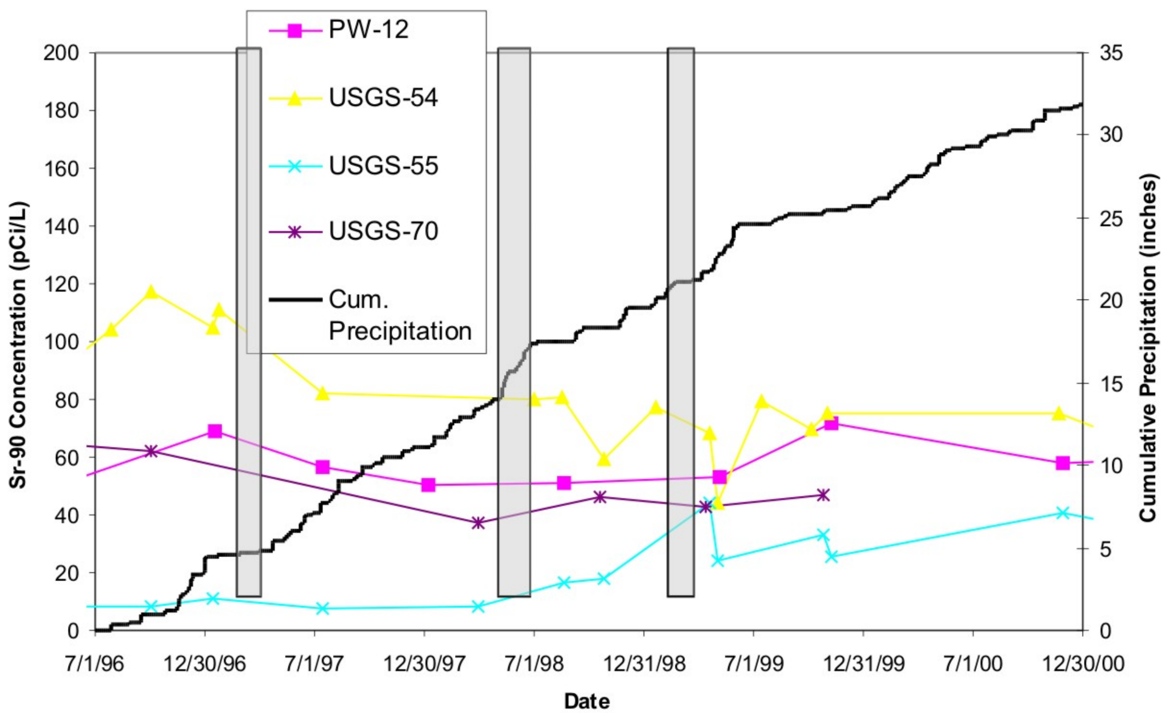

Figure 45. Cumulative precipitation at the Central Facilities Area and strontium-90 concentrations in four perched-water wells.

NOTE: *Shaded intervals indicate unusually moist periods.*

6.2.4 Variations in Recharge from Unidentified Manmade Sources

Leaks from TRA facility infrastructure piping might potentially provide a source of recharge to perched-water bodies at TRA. The water budget presented in Section 7 indicates that nearly all of the water pumped from SRPA production wells for facility use is accounted for in water discharges to the infiltration ponds. However, facility records for several years indicate that more water was discharged to the ponds than was pumped from the aquifer. Uncertainties in the water-budget method probably are derived from measurement errors. These uncertainties limit the capability to quantify leaks from unidentified sources.

Because the TRA water budget is nearly balanced and the resolution of the water balance method is limited by the accuracy of facility gauging and record keeping, it is not possible to adequately determine the presence of infrastructure leaks unless those leaks account for a significant fraction of annual water use. For instance, 1% of annual facility water use during the period of record presented in Section 7 represents 7.8 million gal of water. Most good flow meters are only accurate to +/- 5% of the measured flow rate. As a result, tens of millions of gallons of water would have to be lost before it was observable above the measurement noise of the system. It is conceivable that an undetected leak within the +/-5% error range of the system could have substantial impacts on flow pathways in the perched-water system.

Analysis of water chemistry data (Section 4) suggests that a substantial fraction of perched water is inconsistent with discharge to the CWP. That water exhibits major ion and isotope geochemical characteristics that are consistent with unprocessed, unevaporated SRPA water. Sources of water with these characteristics could include pressurized fire water lines, potable water lines, belowgrade irrigation lines, or other SRPA water lines that feed TRA industrial processes. Water returns from irrigation are expected to have a unique isotopic signature as a result of evaporation and are excluded as a major contributor of the apparent SRPA-derived perched water.

Because the location and quantity of water discharged along any unidentified manmade sources are unknown, the effects on the deep perched zone beneath the TRA are unclear. Depending on the quantity of water that leaked from the source, perched-water levels may or may not be affected by changes in discharge from the source. In cases where a small volume of water is leaked, only the local-scale hydraulics of the perched-water body could be affected. However, as pointed out in Section 5, the effects on concentrations could be significant if the leak occurs near a residual source. For small leaks away from residual sources, the resultant changes to perched-water chemistry will be similar to the changes observed for the mechanisms presented in Sections 6.3.2.2 and 6.3.3 and will be difficult to attribute to this mechanism.

For large leaks, the large-scale shape and hydraulics of the perched aquifer might change. In addition, the location and quantity of water leaked from the unidentified sources could influence flow paths from the contaminant sources to the deep perched zone. These large-scale changes might affect perched-water quality in much the same way as variability in CWP discharge affects water quality (see Section 6.3.2.1).

6.2.5 Lateral Flux from the Big Lost River

The Big Lost River has been considered a potential source of perched water near TRA. However, the river is approximately 4,000 ft from the southeast corner of the TRA fence line on years that it flows. Some evidence has been presented that water levels in two perched-water wells located southeast of TRA (USGS-71 and USGS-62) are affected by exceptionally high flows in the Big Lost River (EG&G Idaho 1991a). These two wells are situated near the southeastern edge of the deep perched-water body (Figure 6). Well USGS-62 was sampled in 1965 during water-level rises that occurred after high

Big Lost River flows and no chemical signature of recharge from the Big Lost River was observed. This lack of chemical response probably is indicative of a hydraulic response without significant mixing of recharged stream flow and existing perched water in the vicinity of the monitoring well.

The Big Lost River flowed frequently during the mid- to late-1990s because of unusually moist weather conditions. During that period, no evidence of water-level increases was observed in Wells USGS-62 or USGS-71. Well USGS-71 was sampled numerous times during that interval and no obvious concentration trends developed for the COCs. Because these wells are the most likely wells to be impacted by lateral flow from the Big Lost River (based on their close proximity to the river) and no obvious and consistent correlation exists with Big Lost River flow, it is not likely that this mechanism is responsible for the increasing and erratic concentration trends observed near the wastewater ponds.

6.2.6 New Leaks of Contamination from Unidentified Sources

Increasing concentration trends could result from new leaks of contamination from unidentified sources. This mechanism is similar to the mechanisms presented in Sections 6.3.2 and 6.3.4 in that the new leaks could affect the hydraulic character of the perched body and influence contaminant advection. However, this mechanism is different in that the other mechanisms do not necessarily provide a new source of contaminants to the perched zone, while this mechanism does.

Warm wastewater was found to leak from the Warm Waste Retention Basin in the early 1970s and continued leaking until 1993. It is estimated that as much as 15 million gal of wastewater per year could have leaked from the basin. That wastewater was part of the same radioactive wastewater stream that discharged the COCs to the WWP. Large leaks of contamination such as this can clearly affect perched-water chemistry. Figure 46 shows a large spike in Sr-90 concentrations in USGS-053 and USGS-054 during the early 1970s that may be explained by leaks from the Warm Waste Retention Basin or a similar unidentified source.

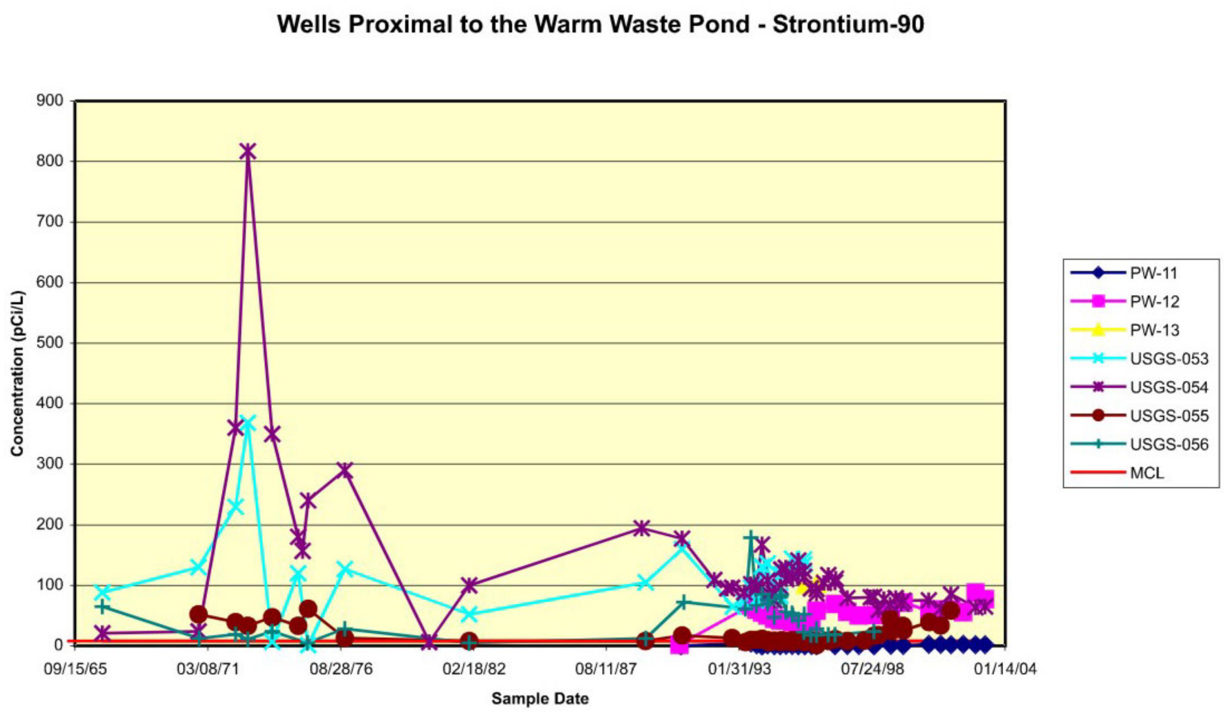

Figure 46. Strontium-90 concentrations in Wells USGS-053 and USGS-054.

The current increasing trend in Sr-90 concentrations and the spike in Co-60 concentrations in Well PW-12 are much lower in magnitude than the spike developed during the early 1970s. If a leak is responsible for the recent observations, then the flux of contaminants from that leak is much smaller than the flux from the source of the early 1970s spike. Section 5 identifies potential sources that could be responsible for the recent Co-60 concentration increase in water collected from PW-12. Strontium-90 concentrations have increased much more gradually than recent Co-60 trends (Figures 41 and 42). If a leak is responsible for the Sr-90 increases, then that leak must be persistent and provide only a small flux of Sr-90 to the perched body; otherwise, a more pronounced increase in Sr-90 concentrations would be expected.

6.3 Alternate Mechanism Identified by Recent Research

Recent large-scale laboratory and small-scale field experiments suggest that the current conceptual model of a slow, diffusive wetting front moving through a fractured rock vadose zone is not valid in many cases. These research investigations used idealized physical models of fracture networks to represent flow in natural systems. The models were easily characterized and monitored, which is a great advantage over the rudimentary understanding gained through many tens of wells on wide spacings at TRA. Through the use of time-lapse photography and sophisticated measuring techniques, several characteristic behaviors began to emerge as common to these systems (Glass et al. 2002a; Wood et al. 2004). These behaviors are not anticipated using conventional predictive, volume-averaged vadose zone numerical models (Fairley, Podgorney, and Wood 2004). Taken at face value, the new research findings suggest that the observed fluctuations in Co-60 and Sr-90 at TRA are to be expected instead of the slow decline predicted using the current conceptual model of idealized flow.

Fracture intersections have been shown to integrate slow, steady, unsaturated flow as water pools above the capillary barrier formed by the intersection (Wood, Nicholl, and Glass 2002; Ji et al. 2004). When the pool height increases sufficiently, the capillary barrier is broken and a large, less frequent discharge occurs below the intersection. Intersections also have been shown to divert flow from one vertical fracture to another (Glass et al. 2002a; Glass et al. 2003; LaViolette et al. 2003). Many fracture intersections interacting in a network commonly cause flow to converge with depth (LaViolette et al. 2003) or cause flow paths to switch between vertical pathways even under steady supply (Glass et al. 2002a). Another common behavior of unsaturated flow in fracture networks is the failure of most to evolve to a steady state; that is, pathways tend to change in discharge rate and/or location over time (Wood et al. 2004). The failure for these systems to converge to a steady state suggests that they are sensitive to small changes in environmental conditions (e.g., minute changes in temperature and barometric pressure).

Although the behavior observed in laboratory experiments has not yet been conclusively documented in a large-scale field problem like the TRA perched-water body, a handful of small-scale field tests suggest that it is likely that they do occur in natural systems (Nicholl and Glass 2002; Glass et al. 2002b; Podgorney et al. 2000; Nativ et al. 1995). The conclusive demonstration of these behaviors at field sites is limited by the difficulty in collecting sufficient monitoring data to track system behavior. Indeed, the fluctuations in Sr-90 and Co-90 observed in some TRA perched-water wells may be the first subtle indications that flow path switching and flow integration do occur in natural systems.

It is possible that changes in the discharge rate to the CWP, fluctuations in barometric pressure, and changes in natural recharge might cause flow path switching in the subsurface beneath TRA. This would expose water to different levels of residual contamination causing concentrations measured in wells to vary in time. Alternatively, slow diffusive flow from the CWP may accumulate above one or more capillary barriers, eventually breaching the barrier and releasing a pulse of water. The pulses of water could create a high level of variability in contaminant levels measured in monitoring wells.

The results are discussed here as one of several plausible explanations for the unexplained fluctuations in Co-60 and Sr-90. There is a strong theoretical basis suggesting that the behavior observed in laboratory experiments may be occurring in the subsurface beneath TRA. More experimental work is needed to clearly correlate field data sets to experimental results. The recent experimental investigations cited above were funded by the DOE Office of Basic Energy Science, Environmental Management Science Program, and Environmental Science Research Alliance.

6.4 Conclusions

Of the six mechanisms considered in Section 6.3, it seems most likely that changes in discharge to the CWP might be the main cause for unexplained increases and decreases of contaminants in perched-water wells. Variations in discharge to the ponds change in response to operational activities, and the location of the water input to the system changes significantly when discharge is cycled from one CWP cell to the other in order to prevent clogging of the pond bottoms by bio-films and sediment. This clearly would change subsurface pathways, exposing water to different levels of residual contamination. Recent research indicates that even if the discharge was constant and to a single pond, flow convergence with depth, flow path switching, and flow integration into pulses would occur. Variations in natural recharge and seasonal input from irrigation further complicate data analysis and probably trigger changes to flow pathways. Complex geochemical relationships between contaminant advection, adsorption, dispersion, and radioactive decay are impossible to sort out under these variable physical flow-field conditions, but undoubtedly affect concentrations in wells, possibly in a nonlinear manner.

It is unlikely that new sources of contamination or undiscovered old sources of contamination are present at TRA, based upon the historical record of concentrations measured during known releases decades ago. The recent unexplained increases and decreases of contamination do not have the same character as the older records. Past system leaks resulted in much larger contamination spikes than the relatively modest increasing trends observed in recent years. The analysis presented in Section 6.3.5 indicates that it is unlikely that lateral flux from the Big Lost River will cause significant changes to the flow field in the vicinity of the TRA fence.

Given the complexity of the subsurface environment and spatial/temporal variability in both the amount and chemistry of recharge sources, short-term variations in concentration levels are to be expected for the perched-water body. These short-term variations are not of concern as long as the trend is seen to decrease again over a maximum period of a few years. Given the current status of scientific understanding regarding flow and transport in unsaturated fractured rock, it is impossible to say what the long-term impacts to aquifer concentrations will be. Currently, there are no numerical simulators that can take advantage of new advances in the experimentally derived conceptual models. We must continue to use the commercially available, volume-averaged diffusive flow and transport models.

Recommendations are as follows:

1. *Continued monitoring of the perched water wells according to the existing groundwater monitoring plan with modifications as approved by DOE, DEQ, and the EPA. It should be realized that not all increasing trends or spikes in contamination pose an immediate or eventual threat to the effectiveness of the remedy and should be evaluated individually to determine the potential of their impact.* Keep current and support ongoing research that focuses on further development of improved conceptual models of unsaturated flow in fractured-rock environments and research that focuses on the development of numerical simulators that can take advantage of new advances in these conceptual models.

7. CONTINUED OPERATION OF THE COLD WASTE POND

7.1 Introduction

The First Five-Year Review Report (DOE-ID 2003) identified continued usage of the CWP beyond 2007 as an issue. At the time of the OU 2-13 ROD (DOE-ID 1997a), it was assumed that the TRA, including the CWP, would be decommissioned in 2007. Under a recent decision (2003) by DOE, the TRA will remain active for at least another 20 years. Continued discharge to the CWP from TRA operations will cause the perched-water systems to persist. This section evaluates contaminant transport to the aquifer under conditions of continued operation through 2027 and beyond. Potential sources of recharge to the perched-water system are evaluated through the development of a water budget for TRA in Section 7.2. A history of modeling efforts leading to the OU 2-13 ROD (DOE-ID 1997a) is presented in Section 7.3, along with an evaluation of the underlying assumptions in light of current understanding. The model is updated to consider continued operations through 2027 in Section 7.4. The findings of this investigation are summarized in Section 7.5, and recommended actions are provided in Section 7.6.

7.2 Water Budget for Sources of Perched Water

A system of perched-water bodies has formed in the vadose zone beneath TRA primarily because of artificial groundwater recharge induced by infiltration of wastewater from unlined ponds at the facility (Figure 1). The presence of perched water beneath TRA might facilitate the downward migration of COCs. The purpose of this section is to evaluate potential sources of recharge to the perched water at TRA. This begins with an accounting of all water used at TRA (Section 7.2.2) and concludes with a discussion of individual sources (Section 7.2.3). The data from this analysis are provided in electronic format in Attachment 3 on the supplemental data CD. A readme file (readme.txt), which describes the electronic data, also can be found on the CD.

7.2.1 Water Balance Accounting

All water used at TRA is currently derived from three aquifer production wells (TRA-01, TRA-03, and TRA-04). Water pumping and discharge data from the TRA facility are recorded in the INEEL Nonradiological Waste Management Information System (INWIMIS) database on a monthly basis. Aquifer pumping data are available for the intervals from January 1975 through December 1984 and from January 1998 through October 2003. A number of anomalously high discharge volumes are recorded in the INWIMIS database prior to January 1983, thus precluding water balance calculations before that date. As a result, only monthly data from January 1983 through December 1984 and January 1998 through October 2003 are used to check the completeness of the water budget at TRA.

In an analysis of the water budget at TRA, water losses are indicated if observed water use is found to be less than the total volume pumped from the SRPA. Such losses could then potentially serve as additional untracked sources of recharge to the perched-water system beneath TRA. During the 1983–1984 period, the average sum of all of the monthly fractional water uses is 101% of the total water pumped. During the 1998–2003 periods, the average sum of the monthly volumetric discharges is 99% of the water pumped. On a monthly basis, the volume pumped might differ from discharge by up to 25% to 30% (perhaps resulting from lag times between water pumpage and disposal). However, the average of the monthly readings indicates that approximately 100% of the pumped water is accounted for by the known discharge sources. If water is lost through system leaks, the fraction of the total pumped water represented by leakage is small compared to other sources (less than 5%).

Table 6 presents the annual fraction of aquifer pumping represented by each discharge source. The fractions were calculated for years in which a full pumping record and discharge record were available. During 4 years in which a full year's worth of pumping and discharge data were available (1975, 1977, 1983, and 2002), the sum of all fractional discharges suggests that nearly 100% of water pumped from the production wells at TRA is accounted for in the discharges. This result is consistent with the results discussed in the previous paragraph for the monthly data.

It is important to note that the average annual volume of water pumped from the SRPA is 780 million gal during the period of record. This means that a 1% water imbalance represents approximately 7.8 million gal of water. System losses are not likely to be noticed unless an imbalance of at least 5% is found; therefore, approximately 40 million gal of water could potentially be leaked to the perched-water system before the water balance accounting method detects the leaks.

7.2.2 Potential Recharge Sources to the Perched-Water System at the Test Reactor Area

As originally designed and installed, two separate wastewater streams were used at TRA: one for sanitary sewage and the second for all other wastewater. The sanitary sewage has always been separate from the other wastewater streams and has always been disposed of at the dedicated sewage lagoons. Over the years, additional segregation of the non-sewage wastewater streams has taken place.

Seven disposal systems in total have been used for long-term wastewater disposal at TRA. The systems include the unlined WWP, the lined Warm Waste Evaporation Pond, the unlined CWP, the unlined Chemical Waste Pond, the Disposal Well, the unlined Sewage Leach Pond, and the lined Sewage Lagoon. Four of these systems—the unlined WWP, unlined CWP, unlined Chemical Waste Pond, and unlined Sewage Leach Pond—have been considered to be sources of recharge water to the perched-water bodies beneath TRA. In addition, several miscellaneous sources might have contributed to recharge, either historically or in the present.

The remainder of this section discusses individual fluid sources that might have contributed to recharge of the perched-water system at TRA. Much of this discussion of historical wastewater streams at TRA in the following sections is excerpted from the *Conceptual Model and Description of Affected Environment for the TRA Warm Waste Pond (Waste Management Unit TRA-03)* (EG&G Idaho 1989). The discharge history for those sources is summarized in Table 7. A historical record of major wastewater discharges is provided in Table 8, and discharge to the four largest contributors is graphed as Figure 4.

7.2.2.1 Warm Waste Pond. The WWP and associated collection system were designed to dispose of radioactive wastewater. The WWP consisted of three unlined cells excavated into Big Lost River gravels. The first cell was excavated in 1952 and had bottom dimensions of 150 × 250 ft. Continued operations resulted in pond bottom plugging, reducing the infiltration capability. Because of the decreased infiltration capacity of the original cell, a second cell was excavated in 1957 with bottom dimensions of 125 × 230 ft. The third cell was excavated in 1964 and had bottom dimensions of 250 × 400 ft.

Originally, all non-sewage wastewater streams were collected in a sump at the southeast corner of TRA until that sump filled to a certain level. At that time, the wastewater was pumped from the sump into the WWP. Before reaching the sump, wastewater containing radionuclide contaminants passed through a Warm Waste Retention Basin intended to allow short-lived radionuclides to decay before final disposal at the WWP.

Table 6. Water balance summary of Test Reactor Area discharges relative to the volume of water pumped from the Snake River Plain Aquifer.

Year	Aquifer Pumpage (gal × 10^6)	CWP	Sewage	Irrigation	Chemical Waste Pond	Radioactive Stream	Desert	Cooling Tower Evaporation	Disposal Well	Sum
		Percentage of Water Pumped from Aquifer								
1975	1,083	—	0.7	1.9	2.4	20.3	22.4	29.0	31.6	108
1976	1,301	—	**	2.2	2.0	15.4	29.1	20.7	35.1	**
1977	981	—	0.9	2.7	2.2	15.0	39.4	‡‡	39.0	99
1978	694	—	1.4	1.9	3.0	18.0	**	‡‡	38.8	**
1979	753	—	1.2	3.3	2.3	9.8	**	‡‡	35.5	**
1980	650	—	1.3	3.5	1.8	8.8	**	‡‡	50.4	**
1981	607	—	1.0	1.6	1.6	9.1	**	‡‡	39.8	**
1982	632	31.3	0.8	2.5	1.4	8.1	**	‡‡	5.9	**
1983	664	37.1	0.9	1.8	1.0	3.9	0.1	55.2	—	99
1984	647	38.2	0.9	2.5	0.9	2.9	—	**	—	**
2002	576	33.8	3.1	9.0	—	1.3	—	52.8	—	100
Average =		35	1.2	3.0	1.9	10.2	22.7	39.4	34.5	

— = Source not in use
** = Incomplete data set for the year
‡‡ = Anomalous readings for CT evaporation during these years (monthly fractions greater than volume of water pumped)
CWP = Cold Waste Pond
WWP = Warm Waste Pond

NOTE: *Based on limited data from the 1970s and 1980s, steam loss and condensate loss are each expected to represent less than 1% of water pumped from the aquifer. The radioactive stream includes wastewater discharged to both the WWP and lined warm waste evaporation pond.*

Table 7. Historical summary of Test Reactor Area wastewater discharge.

Year	Event
1952	Discharge to the WWP and Sewage Leach Pond commences.
1960	The USGS-53 receives wastewater from an unknown source from November 1960 through January 1962.
1962	Chemical Waste Pond is brought online in November 1962.
1963	The USGS-53 receives wastewater from an unknown source between June and August of 1963.
	Discharge line leading to WWP breaks and discharges wastewater to surface 250 ft west of WWP. Leak is repaired.
1964	Aquifer Disposal Well begins receiving waste in November 1964.
	Only warm waste types are disposed of to the WWP. The cold fractions of the waste stream have been diverted to the Disposal Well and Chemical Waste Pond.
Early 1970s	Warm Waste Retention Basin is known to be leaking. The basin continued leaking at a minimum rate of 30 gpm until it was taken offline in 1993.
1982	The CWP is brought online.
	Discharge to the Disposal Well ceases.
1993	Discharge to the WWP ceases.
	Warm waste is diverted to the Lined Evaporation Pond.
1995	Discharge to the Sewage Leach Pond ceases.
	Sewage stream is diverted to lined sewage lagoons.
1999	Chemical Waste Pond is taken offline.

CWP = Cold Waste Pond
USGS = United States Geological Survey
WWP = Warm Waste Pond

Table 8. Annual discharge volumes of potentially important Test Reactor Area perched-water sources.

Year	Warm Waste Pond	Chemical Waste Pond	Irrigation	Cold Waste Pond	Sewage Leach Pond
(Discharge in millions of gal per year)					
1952	75	—	**	—	**
1953	75	—	**	—	**
1954	75	—	**	—	**
1955	97	—	**	—	**
1956	94	—	**	—	**
1957	107	—	**	—	**
1958	266	—	**	—	**

Table 8. (continued).

Year	Warm Waste Pond	Chemical Waste Pond	Irrigation	Cold Waste Pond	Sewage Leach Pond
1959	232	—	**	—	**
1960	221	—	**	—	**
1961	232	—	**	—	**
1962	283	—	**	—	**
1963	202	45	**	—	**
1964	172	45	**	—	**
1965	146	45	**	—	**
1966	130	37	**	—	**
1967	181	45	**	—	**
1968	188	47	**	—	**
1969	279	45	**	—	**
1970	281	45	**	—	**
1971	191	46	**	—	9.3
1972	217	73	**	—	10
1973	269	31	**	—	9.4
1974	246	31	**	—	8.2
1975	220	26	20	—	7.5
1976	200	26	28	—	8.7
1977	147	22	26	—	9.3
1978	125	20	13	—	9.4
1979	74	17	25	—	8.7
1980	57	12	23	—	8.2
1981	55	9.3	10	—	5.8
1982	51	8.9	16	203	5.1
1983	26	6.7	12	238	6.1
1984	19	5.8	16	248	6.0
1985	20	6.0	12	222	7.2
1986	25	6.3	13	272	8.5
1987	19	5.5	26	178	6.8
1988	18	4.2	54	224	7.4
1989	23	7.7	54	294	8.2

Table 8. (continued).

Year	Warm Waste Pond	Chemical Waste Pond	Irrigation	Cold Waste Pond	Sewage Leach Pond
1990	20	7.5	51	254	8.6
1991	29	8.6	75	206	13
1992	23	8.5	125	143	13
1993	17	6.4	72	250	14
1994	—	6.0	47	214	33
1995	—	6.4	**	169	22
1996	—	5.6	60	162	—
1997	—	5.9	**	172	—
1998	—	4.5	65	157	—
1999	—	0.96	—	36	—
2000	—	—	23	181	—
2001	—	—	34	152	—
2002	—	—	52	195	—

— = Source did not exist or was not included in recent utility reports.
** = Data were not recorded in INWIMIS database; volume either zero or unknown.
INWIMIS = INEEL Nonradiological Waste Management Information System

From 1952 through 1962, the WWP received all non-sewage wastewater. In 1962, the fraction of wastewater from the Demineralization Plant was diverted from the WWP to the newly constructed Chemical Waste Pond. By 1964, only wastewater containing radionuclides was disposed of in the WWP. All three cells were taken offline and remediated in late 1993. At that time, the warm wastewater stream was diverted to the newly constructed, lined Warm Waste Evaporation Pond.

As presented in Table 8, between 17 million and 283 million gal of wastewater was disposed of annually in the WWP from 1952 until the pond's closure in 1993. Before the addition of the CWP in 1982, discharges to the WWP represented the primary source of water to the TRA perched-water system.

7.2.2.2 Test Reactor Area Disposal Well. The TRA disposal well was constructed to dispose of nonradioactive wastewater from cooling tower blowdown directly to the SRPA. Wastewater containing chromate was discharged to the TRA disposal well beginning in November 1964 and ending with the construction of the CWP in 1982. The well presently is used as a monitoring well and is screened over several intervals between 512 and 1,267 ft bls.

7.2.2.3 Cold Waste Pond. The CWP was brought online in 1982 and has been used since that time as a replacement for the Disposal Well. The pond consists of two cells, each 150 × 400 ft. Water disposed of at the CWP primarily originates from cooling tower blow-down, air-conditioning units, secondary system drains, floor drains, and other nonradioactive drains throughout TRA (DOE-ID 2003). Historically, only one of the two cells has been used at a time with rotation on an annual basis.

Wastewater discharge to the CWP has been the largest contributor to the TRA perched-water system since it came online in 1982. As presented in Table 8, between 36 million and 294 million gal of wastewater has been disposed of each year in the CWP. The CWP is projected to receive approximately 300 million gal of wastewater per year through at least 2024.[d]

7.2.2.4 Sewage Leach Pond. The Sewage Leach Pond consisted of cells excavated in 1950 and 1965 that received discharge from sanitary sewer drains. Process knowledge indicates that effluent was limited to domestic sewage (DOE-ID 2003). Low-level radionuclides were detected in the 1950 cell, but have been attributed to airborne dust accumulation in the pond. Sewage discharge rates to the pond are available beginning in 1971.

Annual discharge to the unlined Sewage Leach Pond varied between 5.1 million and 33 million gal of wastewater from 1971 to its closure in 1995 (Table 8). The Lined Sewage Lagoon was brought online with the decommissioning of the Sewage Leach Pond.

7.2.2.5 Chemical Waste Pond. The Chemical Waste Pond was excavated in 1962 and was first used in November of that year to dispose of wastewater from ion exchange columns and water softeners. The pit was unlined and had bottom dimensions of 170 × 170 ft. Wastewater discharged to the Chemical Waste Pond contained sulfuric acid, sodium hydroxide, and sodium chloride. The pond was taken out of service in 1999 (DOE-ID 2003).

Table 8 presents wastewater volumes discharged to the Chemical Waste Pond from the time of its construction in 1964 to its final remediation in 1999. Annual volumes of disposed water ranged from approximately 1 million gal in 1999 to a maximum of 73 million gal in 1972.

7.2.2.6 Other Potential Perched-Water Sources and Sinks. Other potential sources and sinks exist that could affect the perched-water system. These sources are either poorly quantified, infrequent, or potentially minor contributors to the perched-water system based on the relative magnitude of their volumetric discharge compared to the major contributors identified in the previous sections.

7.2.2.6.1 Lawn Irrigation—Lawn irrigation presents a potential source of perched water. Most grassy areas inside the TRA fence that would require irrigation during the spring through fall months are located in the central, northern, and western portions of the facility. The annual volume of irrigation discharge ranged from 10 million gal in 1981 to 125 million gal in 1992 (Table 8).

7.2.2.6.2 Known Leaks— Several leaks and/or unplanned discharges are known to have occurred since the commencement of operations at TRA. They include the following:

- Since the early 1970s, it was known that the Warm Waste Retention Basin leaked, although it is not known when the leak started. The exact rate of leakage is unknown, although 30 gpm is a minimum estimate. At 30 gpm, leakage would have exceeded 15 million gal of wastewater each year. The leaking portion of the basin was taken offline in 1993, coincident with the construction of the new lined Warm Waste Evaporation Pond.

- In August of 1963, a wastewater line leading to the WWP broke and discharged liquids to the surface approximately 250 ft west of the pond. A small perched-water zone that formed in surficial alluvial deposits beneath this leak diminished after the leak was repaired.

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d. Personal communication with TRA Facility Operations, April 2004.

7.2.2.6.3 Wastewater Discharge to USGS-53—Wastewater from an unknown source was disposed of in the USGS-53 perched-water well from November 1960 through January 1962 and June 1963 through August 1963. Well USGS-53 is completed to a depth of 90 ft bgs in the deep perched-water zone. While the source of wastewater disposed of in the well is unknown, injection rates have been estimated at about 148,000 gal/day. Over the period of operation presented above, it is estimated that a total of 220 million gal of wastewater was discharged directly to the deep perched-water zone through USGS-53 (Dames and Moore 1992a).

7.2.2.6.4 Desert—During the 1970s, discharge to locations designated as "Desert" and "Desert Depression" in the INWIMIS database are recorded. The location(s) of these discharge point(s) are unknown and may or may not have had a significant impact on the perched-water system at TRA. Over 200 million gal of water was discharged to this location (or locations) during the mid-1970s (see Attachment 3, provided on the supplemental data CD).

7.2.2.6.5 Steam and Condensate Loss—Steam and condensate loss are listed in the INWIMIS database during a portion of the 1970s and 1980s. During that time, monthly losses from these sources account for less than 1% each of the total water pumped from the aquifer. Based on available aquifer pumping data, 1% of the pumped volume represents less than 10 million gal of water per year. Some of this loss can potentially reach the perched-water system through direct discharge to the subsurface.

7.2.2.6.6 Big Lost River—The Big Lost River has been considered a potential source of perched water near TRA. However, the river channel is approximately 4,000 ft from the southeast corner of the TRA fence line. Some evidence has been presented that water levels in two perched-water wells located southeast of TRA (USGS-71 and USGS-62) are affected by exceptionally high flows in the Big Lost River (EG&G Idaho 1991a). No chemical signature of recharge from the Big Lost River was observed at USGS-62 during water-level rises that occurred after high Big Lost River flows in 1965.

Because it is difficult to determine when TRA water levels might be affected by Big Lost River flow, let alone how much water might be contributed to the perched-water system southeast of the facility, this source is an unknown. However, the river's lateral and vertical distance from the TRA fence line and the vicinity of the CWP make it unlikely that its effect on the perched-water system would be significant. However, the mounding height of the perched water can potentially be used for further analysis and provide a definitive determination.

7.2.2.6.7 Precipitation—Average annual precipitation in the southwestern portion of the INEEL is 8.7 in. (DOE-ID 2003). Most of this precipitation falls as snow during the winter and spring months and subsequently melts as warmer spring temperatures arrive. The pre-ROD perched-water system model implemented a single, natural (i.e., precipitation-induced) infiltration rate of 5.92 in./year over the entire simulation period, which equates to over 100 million gal/yr spread over the greater area of TRA. This infiltration rate was insufficient to sustain perched-water bodies after manmade discharges were shut off in the calibrated model.

7.2.2.6.8 Evaporation—The discharge volumes reflected in Tables 6 and 7 are gross discharge volumes and do not reflect the effects of evaporation. Evaporation from small lakes in southeast Idaho (a surrogate for the TRA wastewater impoundments) is on the order of 32 to 36 in./yr (Linsley, Kohler, and Paulhus 1982). For the periods during which at least two of the three WWP cells were filled, this evaporation rate (and the estimate area from which evaporation would occur) would have accounted for approximately 1 to 8% of the total warm waste discharge volume (EG&G Idaho 1989). Similar calculations could be conducted to determine the percentage of water lost to evaporation at other TRA water impoundments based on their submerged surface area and the evaporation rate presented previously. The pre-ROD perched-water system model assumes that the evaporation rate is 10% of pond discharge, regardless of pond surface area (Dames and Moore 1992a).

7.3 Modeling History and Evaluation of Perched-Water System Model Assumptions

Modeling of the perched-water system and underlying SRPA has been conducted to examine in a predictive manner the nature and extent of the contamination in the subsurface at TRA. Additional modeling is required, because the pre-ROD model assumed that TRA would cease operations in 2007; however, this assumption is no longer valid. The general approach for this modeling effort involves the use of two numerical codes: the original model updated to extend the period of operation to 2024 and the use of a modern, commercially available, state-of-the art numerical code. This approach was taken to (1) provide a higher level of assurance for predictions and (2) provide redundancy for the calculations made by the original code, which is now somewhat antiquated and out of date. This section describes the history and rationale of the modeling efforts (Section 7.3.1) as well as the simulation code used (Section 7.3.2). A presentation and discussion of the assumptions used in the modeling effort also are provided (Sections 7.3.3 and 7.3.4, respectively).

7.3.1 Modeling History and Rationale

The TRA perched-water system model was developed in 1991 to provide exposure-point concentrations for the TRA human health and ecological risk assessment and to predict both vadose zone moisture conditions and contaminant concentrations in the SRPA. The results of those simulations were used to evaluate remedial action alternatives under the CERCLA process.

The OU 2-12 ROD (DOE-ID 1992) identifies procedures that are being employed to ensure that contaminant concentrations are protective of human health and the environment. The OU 2-12 ROD called for no remedial action, because the human health and ecological risk assessments determined that conditions at the site pose no unacceptable risks to human health or the environment for expected current and future site users (i.e., users in the year 2115). The No Remedial Action decision was based on the requirement that groundwater monitoring be conducted to verify that aquifer contaminant concentration trends match the results predicted with the perched-water system model. In addition, the decision was based on the assumptions that operations at TRA will continue through at least 2007 and that the WWP, which was the major source of contamination to the aquifer, was to be replaced with a lined pond in 1993.

The OU 2-13 ROD (DOE-ID 1997a) reaffirms the OU 2-12 ROD (DOE-ID 1992) and describes activities that are necessary for protection of human health, as follows: "Groundwater monitoring will be conducted to verify that contaminant concentration trends follow those predicted by the groundwater model. Computer modeling shows that through natural radioactive decay, natural attenuation, and dispersion, contaminants in the groundwater will steadily decrease to acceptable levels within the next 20 years, which is consistent with the time of continued operations at the TRA."

The First Five-Year Review Report (DOE-ID 2003) and the OU 2-12 ROD (DOE-ID 1992) state that the perched-water system model predicted that tritium and chromium concentrations were expected to fall below their MCLs by 2004 and 2016, respectively.

In order to evaluate the ability of the existing perched-water system model to accurately predict contaminant concentrations in the SRPA beneath TRA, it is necessary to evaluate the assumptions of that model in light of current understanding of the TRA conceptual model. Section 7.2.2 presents the assumptions inherent in the existing perched-water system model and discusses their potential impact on the prediction of contaminant concentrations in the SRPA.

7.3.2 Simulation Code Selection

The TARGET-2DU computer code, Version 4.3 (Dames and Moore 1985; Dames and Moore 1995) was used to simulate flow and transport beneath TRA in support of a human health risk assessment (Dames and Moore 1992b). The TARGET-2DU computer code is one component of the TARGET suite of modeling software, which also can be used to simulate other flow conditions such as three-dimensional, saturated flow- and density-coupled flow using a grid-centered, finite-difference approach. The rationale used to select TARGET is documented in Dames and Moore (1992a), who state that TARGET is peer-reviewed, commercially available, and technically capable of simulating the interaction of the perched-water system and SRPA at TRA.

7.3.3 Perched-Water System Model Assumptions

The existing perched-water system model is a two-dimensional, cross-sectional model oriented parallel to the local hydraulic gradient in the SRPA. While flow directions are variable both spatially and temporally, the average direction of groundwater flow in the SRPA below TRA is to the southwest (DOE-ID 2003). Because the primary sources of contaminants to the subsurface, the WWP, and CWP are aligned along a northeast-southwest oriented line and sufficient wells exist along that line to constrain the geologic framework, the two-dimensional model was oriented to include those features. The cross-section discretized into the model is approximately 2 mi long and 700 ft deep, approximately centered on the CWP (as shown on Figure 5-3 in Dames and Moore [1992b]).

The assumptions listed below were made by Dames and Moore (1992b) during the development of the perched-water system model:

1. The perched-water system can be represented by a two-dimensional slice model.

2. Flow is vertical except in the shallow and deep-perched zones.

3. The perched-water system is an equivalent porous medium.

4. Individual layers are homogeneous.

5. Basalt is anisotropic with vertical hydraulic conductivity (K_v) greater than horizontal hydraulic conductivity (K_h). For sediments, K_v is less than K_h.

6. Moisture-characteristic curves for fractured tuff are applicable to the unsaturated zone at TRA.

7. Active thickness of the SRPA is 250 ft.

8. Interaction of the perched-water system and the Big Lost River is negligible.

9. The WWP and CWP are the primary sources of perched water.

10. Chromium injected to USGS-53 can be modeled as going into the WWP.

11. Sources of water to the subsurface include known wastewater sources and precipitation. No other potential sources of groundwater recharge are included in the model.

12. Discharge to the CWP would end in 2007 with closure of the TRA facility.

13. Discharge to the CWP will continue from 1990 through 2007 at the rate observed in 1990 (approximately 250 million gal/yr).

7.3.4 Discussion of Model Assumptions

The modeling assumptions presented in the previous section were discussed and defended at the time of model development (Dames and Moore 1992b). This section presents a summary of the modeling assumption discussion presented by Dames and Moore (1992b) and provides some additional insight into the model assumptions gained during the model update process to be discussed in Section 7.4.

Assumption 1 implies that the geologic and hydraulic setting in the plane of the cross-section is representative of the setting outside this plane. If there is significant variation outside the plane of the model, this assumption may be violated. Factors that might influence the physical setting outside the plane of the model include the presence of recharge sources and different stratigraphy outside the model cross-section.

Assumption 2 implies gravity-driven flow conditions to exist in areas of the vadose zone outside of the perched-water bodies. Within the perched-water bodies, as well as in unsaturated areas above the perched water, the flow of infiltrating water includes a lateral component.

Assumption 3 was considered valid because the point of interest (the SRPA) is relatively far from the source and breakthrough curves for contaminants that resemble those that would be obtained for a porous medium.

Assumption 4 was necessary because, even though heterogeneity is know to exist, there is a lack of data available to define it within the individual model layers. Dames and Moore (1992b) stated that this assumption is valid if the model recreates observed conditions since heterogeneity in the layers is known to exist.

The relationship of K_v to K_h in basalt presented in Assumption 5 is contrary to the accepted conceptual model of flow in fractured basalt, which dictates that rubble zones allow rapid migration of fluids in the horizontal plane. However, the relationship used in the model is consistent with the concept of higher K in the direction of flow (i.e., the stratigraphic control on the flow is different depending on the direction of flow).

Assumption 6 was made because characteristic curves for fractured basalt were unavailable at the time of the 1992 modeling.

Assumption 7 was based upon the *Conceptual Model and Description of the Affected Environment for the TRA Warm Waste Pond (Waste Management Unit TRA-03)* (EG&G Idaho 1989), which discussed the thickness of the SRPA.

Assumption 8 was based upon several observations, specifically: (1) there is no consistent pattern of perched-water-level rises attributable to Big Lost River infiltration; (2) no perched-water-level rises west, northwest, or north of the CWP and WWP were observed as a result of Big Lost River infiltration; (3) the bulk of contaminated perched water is northwest of the ponds; and (4) the INEEL flood diversion system precludes the possibility of large amounts of infiltration from the Big Lost River as a result of flood inundation.

Assumption 9 implies that error associated with lumping contaminant sources into the two ponds should be minimal.

Assumption 10 essentially results in an underestimation of travel time to the aquifer and overestimation of sorption for the water injected at USGS-53. These elements were not considered important by Dames and Moore (1992b) because of the long simulation period and short travel time to the SRPA.

Assumption 11 was not explicitly stated in the original modeling report. It amounts to effectively neglecting the infiltration of irrigation water applied to lawns inside the TRA fence. The validity of this assumption can be questioned.

Assumptions 12 and 13 are no longer valid based on current knowledge of TRA facility operations. The facility will not be closed by 2007. Under a recent decision by DOE, TRA will remain active for at least another 20 years (DOE-ID 2003).

Dames and Moore (1992b) argued that the relatively good calibration of the model to chromium and tritium concentrations and observed heads validates Assumptions 1–10. Assumptions 11–13 were not expected to have had a significant impact on the calibration of the model. However, those assumptions might affect predicted future concentrations of contaminants in the SRPA.

During a qualitative assessment of model sensitivity to various input parameters, Dames and Moore (1992b) found that the model is not very sensitive to natural infiltration (precipitation) rate changes and that the natural infiltration rate used in the model is conservative (i.e., high). A similar sensitivity assessment could be conducted to determine the model sensitivity to infiltration of irrigation water. If the model is relatively insensitive to infiltration rates that approximate those resulting from lawn irrigation, then Assumption 11 could be considered valid. If the effects of simulated lawn irrigation are large, which would be indicated by significant perched-water body formation from irrigation water alone, then Assumption 11 would be invalid and Assumption 1 would need to be reevaluated. Because the irrigated portions of the facility do not lie on the model transect, the applicability of a two-dimensional model (as assumed in Assumption 1) would be questionable if infiltrating irrigation water has large effects on the perched-water system.

7.4 Update of the Pre-Record of Decision Perched-Water System Model

This section describes the activities involved with updating and revising the pre-ROD perched-water system numerical model as well as comparing the TARGET model with the commercially available code TOUGH2 (Section 7.4.1). The updated TARGET modeling results are compared with the pre-ROD predictions as well as the TOUGH2 results in Section 7.4.2.

7.4.1 Model Update

The pre-ROD perched-water system model implemented using TARGET assumed that the TRA facility would close in 2007 and discharge to the CWP would cease at that time. The mission at TRA has been extended since the original model was developed, and the CWP is expected to receive wastewater at current levels through at least 2024. The pre-ROD model was updated to consider the effects of this operational change. In addition, the model has been further extended to evaluate the impact of CWP discharge beyond 2024.

This section identifies the COCs for which updated simulations were conducted, describes the changes made to the original perched-water system model, and presents updated simulation results.

7.4.1.1 Extended Period for Existence of Perched-Water Sources. The original perched-water system model simulated discharge to the CWP through 2007. However, current TRA facility plans dictate that the CWP will continue to receive wastewater at an estimated rate of 25 million gal per month (300 million gal/yr) through at least 2024 (personal communication, see footnote d). This change was implemented in the updated TARGET model by adding CWP discharge at 270 million gal/yr (300 million gal/yr minus 10% for evaporation) to the model from 2003 through 2024. During the development of the original model, it was assumed that evaporation equaled 10% of the volume of water discharged to the wastewater ponds (Dames and Moore 1992a). In addition, the CWP discharge volumes used in the original model for the period from 1990 through 2002 were revised to reflect the annual discharges measured by TRA Facility Operations as presented in Table 8 (minus 10% for evaporation). Because the site COCs were not present in CWP wastewater, no changes to contaminant source terms were necessary.

7.4.1.2 Updated Contaminants of Concern. At the time the pre-ROD TARGET model was developed, limited concentration data were available for analytes other than tritium and Cr. As a result, the contaminant transport portion of the model was calibrated solely to these compounds (Dames and Moore 1992b). Simulations were originally carried out using the TARGET model for a number of potential COCs, including Am-241, As, Be, Cd, Cs-137, Cr, Co-60, F, Pb, Mn, Sr-90, tritium, and Hg.

During the data analysis for the First Five-Year Review Report (DOE-ID 2003), only Cr, tritium, Co-60, and Sr-90 were reported to have a significant impact on perched water and the aquifer. As a result, updated simulations were only carried out for these four COCs. The results of the updated simulations are presented in Section 7.4.2.

7.4.1.3 Revisions to the TARGET Model. Despite the lack of changes to contaminant source terms, two omissions that affect simulated contaminant concentrations were discovered during the model update process; radioactive decay of Co-60 and Sr-90 was not implemented in the original perched-water system model. The Co-60 and Sr-90 half-lives of 5.20 and 28.0 years, respectively, were implemented in the updated model. Because any previous risk calculations were conducted using simulated Co-60 and Sr-90 concentrations that were unaffected by radioactive decay, those concentrations can be considered conservative.

7.4.1.4 TOUGH2 Modeling. Simulations of vadose zone flow and transport also were carried out using the TOUGH2 numerical simulator (Pruess 1991). The setup and conditions of the TARGET simulations were duplicated (i.e., modeled domain, boundary conditions, grid size, and parameters) to the degree possible with the TOUGH2 code. These efforts were conducted in order to compare the results with a modern, well-documented, and widely accepted commercially available code. The objective of these simulations was to provide a basis of comparison with the results of the updated TARGET modeling (with the CWP operational until 2024). Development of an independent numerical model was beyond the scope of this task.

Several modifications to the simulated domain and parameters had to be made to implement the simulations with the TOUGH2 code. The TARGET code uses a unique formulation for calculating the relative permeability and capillary pressure functions for vadose zone flow that are not available in the TOUGH2 code. Therefore, the Leverett function (1941) was used to calculate the relative capillary pressure in the TOUGH2 simulations, while the van Genuchten model (1980) was used to determine the relative permeability. Data points were extracted from the plots of pressure head vs. moisture content presented by Dames and Moore (1992a, Figure 5-5) and used in the computer code RETC (van Genuchten, Leij, and Yates 1991) to determine parameters for the van Genuchten's relative permeability model.

As was performed in the TARGET modeling, the model results for chromium and tritium were first compared with the monitoring data in USGS-65. In order to better match the field observations, several minor modifications were made to the parameter values used in the TARGET simulations: (1) the vertical hydraulic conductivity in the SRPA was reduced by a factor of 8, (2) van Genuchten relative permeability model “m” parameter in the alluvium was increased from 0.15 to 0.25, and (3) K_d values for the lower interbed were increased by 30%.

7.4.1.5 Model Comparison. The numerical modeling codes used in this assessment vary significantly in their implementation for a particular problem. The TARGET code consists of four independent computer codes for (1) two-dimensional, vertically integrated, confined/unconfined, transient groundwater flow, and solute transport; (2) two-dimensional vertically oriented (cross-section), variably saturated, density-coupled, transient groundwater flow, and solute transport (used for the TRA simulations); (3) multilayer (variable layer thickness), confined/unconfined, transient groundwater flow, and solute transport; and (4) three-dimensional, saturated, density-coupled, transient groundwater flow, and solute transport. The TOUGH2 code consists of a single, primary executable routine that is linked to one of a number of Equation of State modules, depending on the problem at hand. The TOUGH2 code can accommodate all of the applications of the four TARGET executables as well as a significant suite of other, more complex applications.

The TARGET code is a proprietary code that is supplied as an executable program by the authors. This is in contrast to TOUGH2, which is a public domain code supplied as basic FORTRAN files that must be compiled and linked by the user. This difference makes application of the codes vary significantly. The TOUGH2 code, owing to the ability to modify the source code and flexibility built into the primary executable, has the capabilities to handle many more situations than the TARGET code, but owing to this ability, many more input variables might be required (that may or may not be available).

For the present application, the TOUGH2 code was made to mimic the TARGET code to the extent possible (i.e., using similar property sets and functions for the simulation of flow and transport). Differences that cannot be reconciled between the two codes, such as the numerical solution schemes (TOUGH2 uses a set of fairly advanced routines that are selectable by the user), grid cell connection averaging, and weighting can account for the differences in the simulation results.

7.4.2 Model Results

Predicted concentrations presented in the OU 2-12 ROD (DOE-ID 1992) and used during the human health and ecological risk assessment represent the maximum concentrations in the top 12.5 ft of the SRPA in the model domain (Dames and Moore 1992a). Maximum concentrations were assumed to occur in that interval, directly beneath the waste ponds, before mixing and dilution in the aquifer could occur. Dames and Moore (1992a) noted that risks calculated using the 12.5-ft screened interval at the top of the aquifer are likely to be conservative, because typical SRPA wells are completed over a larger saturated (40- to 100-ft) interval, which is expected to yield a more diluted sample than the predictions would indicate.

The TOUGH2 modeling results were comparable with those obtained with the TARGET modeling. The TOUGH2 simulation results in general showed lower maximum concentrations of the four COCs in the SRPA. With the exception of tritium, the TOUGH2 simulations showed concentrations returning to near baseline concentrations in the aquifer earlier than in the TARGET simulations.

Contaminant concentrations predicted in the upper 12.5 ft of the aquifer have been presented in previous documents and represent maximum concentrations in the aquifer; therefore, results presented in this section are also derived from that interval. The following sections present the results of the original

(CWP discharge until 2007) and updated (CWP discharge through 2024) model simulations for the four COCs as well as the TOUGH2 modeling results.

7.4.2.1 Chromium. The original TARGET perched-water system model predicted that Cr concentrations would fall below MCLs by 2016. The updated model predicts that Cr concentrations in the SRPA will remain above MCLs until 2034, assuming that discharge to the CWP ceases in 2024. As portrayed in Figure 47a, the TARGET model predicts a sharp decreasing trend in Cr concentrations that begins at the time CWP discharge is simulated to end in the updated model. This result can be explained by the expected reduction of Cr flux from the vadose zone to the aquifer after 2024, when the only source of groundwater recharge is precipitation.

The TOUGH2 model predicted lower concentrations of chromium in the SRPA under conditions of the CWP being operational until 2024 (Figure 47b) with concentrations falling below the MCLs in approximately 2022. The concentration of chromium is predicted to drop rapidly with cessation of flow into the CWP.

7.4.2.2 Tritium. The results of the pre-ROD and updated TARGET modeling (2024 CWP operations) of tritium show very little change in the predicted concentration in the SRPA. All the TARGET models predict that tritium concentrations are near MCLs at the present time (2004). Because of the radioactive decay of this relatively short-lived isotope, tritium concentrations are expected to continue to fall and remain below MCLs for the entire simulated period (Figure 48a). Tritium concentrations are not predicted to rise above MCLs in the updated perched-water system model. The TOUGH2 modeling predicted tritium concentrations to fall below MCLs in the year 2018 (see Figure 48b).

7.4.2.3 Cobalt-60. The effects of both the cessation of discharge to the CWP and radioactive decay are evident in the pre-ROD and updated TARGET simulation shown on Figure 49a. Cobalt-60 is more highly retarded in the vadose zone than more conservative species such as Cr or tritium. Therefore, it reaches peak concentrations after the wastewater discharge period simulated in the pre-ROD and updated TARGET models. Once the flux of Co-60 into the aquifer is reduced, radioactive decay plays a more significant role in defining the concentration trend. This COC is not predicted to exceed MCLs in the SRPA at any time under the extended TRA operational scenario. The results of the TOUGH2 modeling show similar trends as the TARGET model; however, the peak concentrations are approximately 25% of those predicted with the TARGET model, as shown on Figure 49b.

7.4.2.4 Strontium-90. The effects of both the cessation of discharge to the CWP and radioactive decay are also evident in the predicted concentration trends for Sr-90 (Figure 50a). As with Co-60, Sr-90 is more highly retarded in the vadose zone than Cr or H-3 and therefore does not reach peak concentrations until after the wastewater discharge period ends in the pre-ROD and updated TARGET models. In the extended model, Sr-90 concentrations reach a peak of ~7.6 pCi/L at approximately 2102 and begin declining trends until the end of the simulation period. This COC is not predicted to exceed MCLs in the SRPA at any time under any of the TRA operational scenarios. Comparison of the TOUGH2 and TARGET modeling (Figure 50b) is similar to the results for Co-60 with the TOUGH2 simulations predicting similar trends but with a maximum concentration of approximately 25% of that in the TARGET simulations.

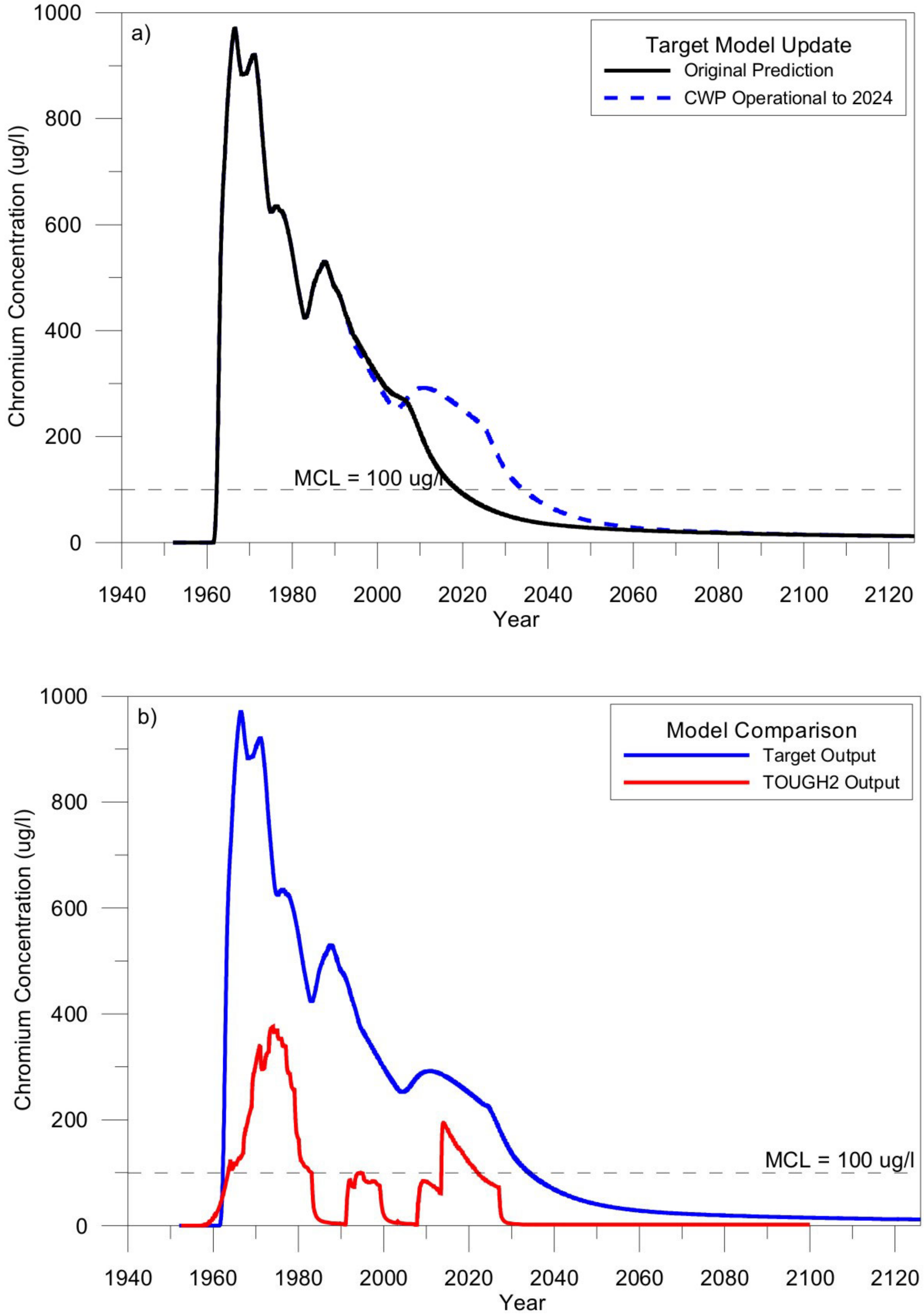

Figure 47. (a) Original and updated TARGET predicted maximum chromium concentration in the Snake River Plain Aquifer and (b) comparison of the TARGET and TOUGH2 results.

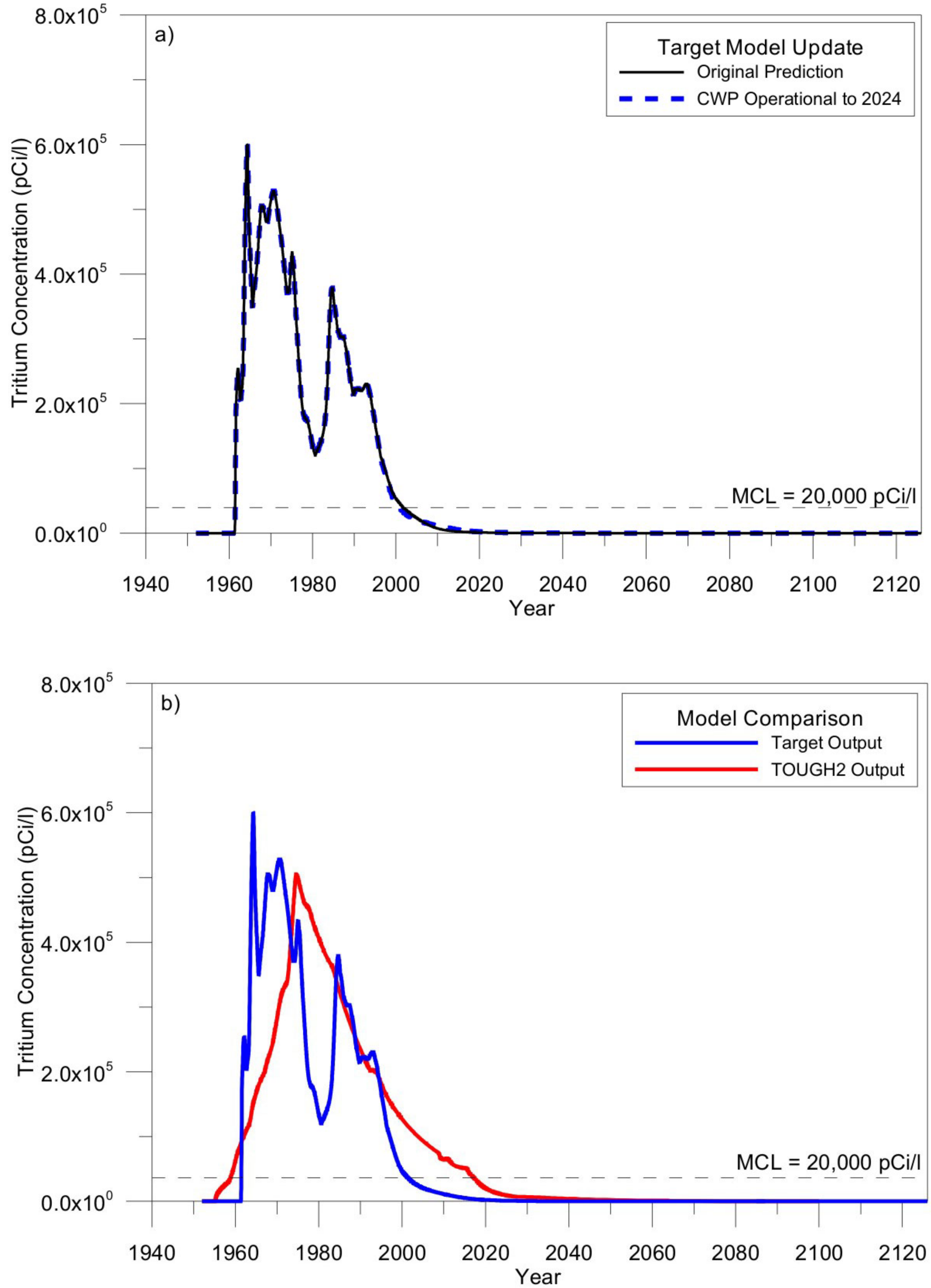

Figure 48. (a) Original and updated TARGET predicted maximum tritium concentration in the Snake River Plain Aquifer and (b) comparison of the TARGET and TOUGH2 results.

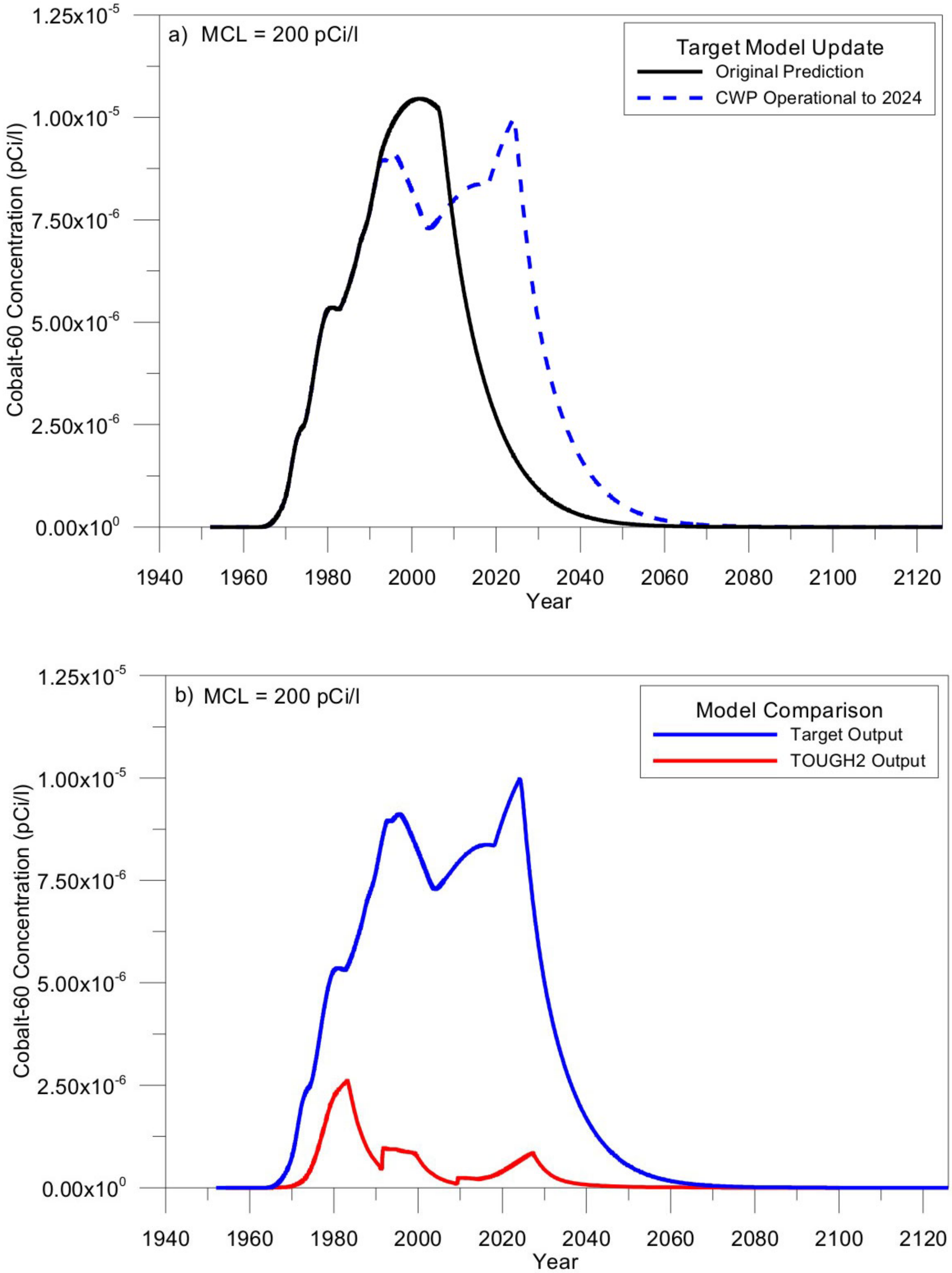

Figure 49. (a) Original and updated TARGET predicted maximum cobalt-60 concentration in the Snake River Plain Aquifer and (b) comparison of the TARGET and TOUGH2 results.

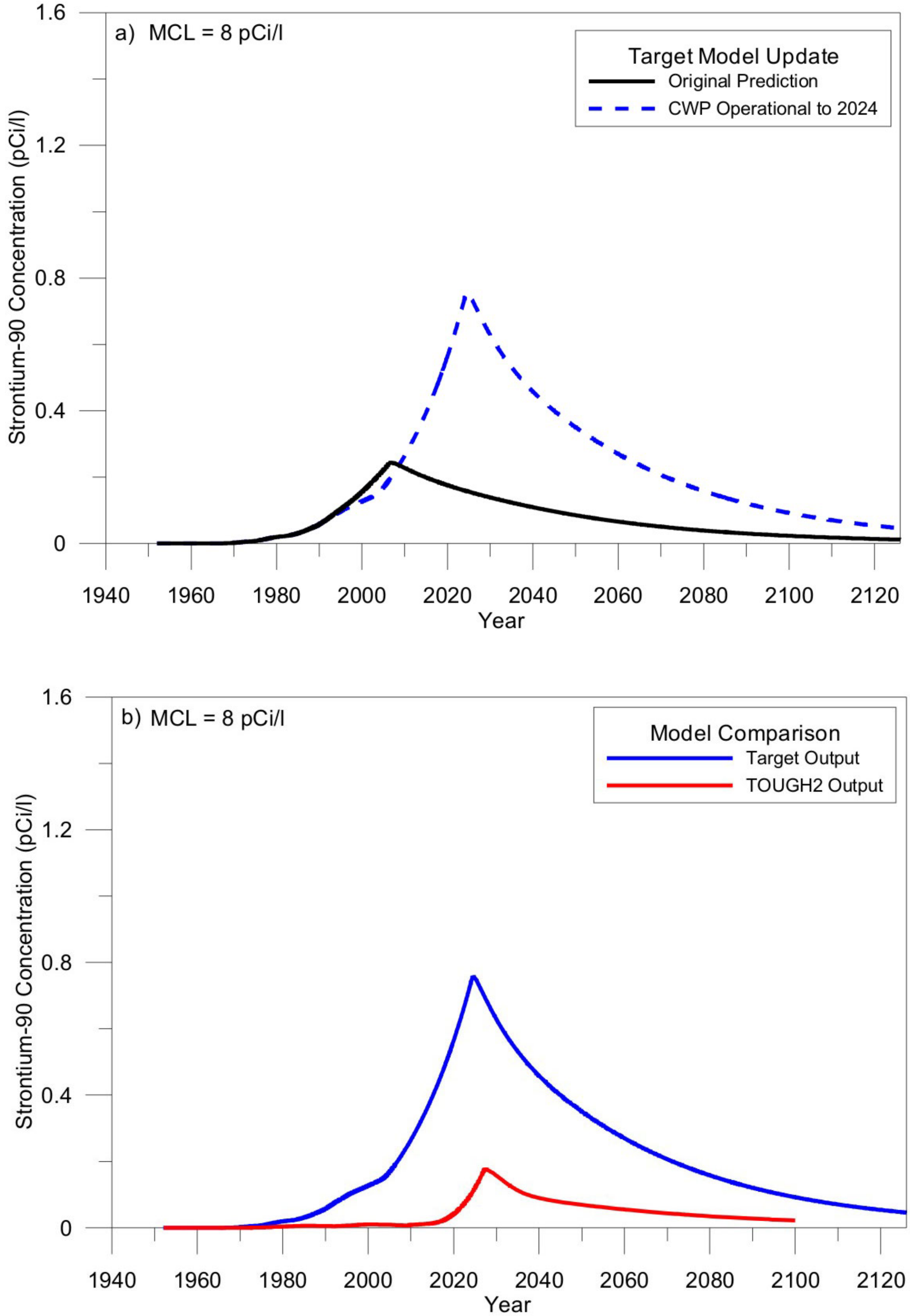

Figure 50. (a) Original and updated TARGET predicted maximum strontium-90 concentration in the Snake River Plain Aquifer and (b) comparison of the TARGET and TOUGH2 results.

7.5 Conclusions and Recommendations

The following recommendations/conclusions can be made following the evaluation of the TRA water budget, modeling assumptions, and updated modeling efforts.

7.5.1 Water Budget for Sources of Perched Water

Four unlined, manmade wastewater impoundment structures have contributed to the perched-water system at TRA at rates of at least 10 million gal each per year; they are: (1) the WWP, (2) the CWP, (3) the Chemical Waste Pond, and (4) the Sewage Leach Pond. Of these, the WWP and CWP have each received more than 200 million gal of wastewater during a single year and are (or were) the primary contributors to the perched-water system. Additional sources of water to the perched-water system are lawn irrigation, precipitation, discharge to Well USGS-53, discharge to an unknown desert location, and known infrastructure leaks.

A water balance consisting of all TRA water-use sources and sinks was conducted in an effort to identify previously unknown water losses. The water balance was nearly complete based on a limited number of years for which data from all sources and sinks were available. No obvious water losses were identified. However, water losses less than approximately 40 million gal/yr are unlikely to be detected based on the resolution of this water balance accounting method.

7.5.2 Perched-Water System Model Assumptions

At least three assumptions used during the development of the existing perched-water system model could affect predicted concentrations. First, the model does not include infiltration of water from irrigated portions of the TRA facility. The importance of this assumption should be evaluated by assessing the sensitivity of the model to infiltration rates that approximate infiltration from the irrigated area. The sensitivity of the model to irrigation water infiltration could affect the assumption that a two-dimensional slice model is appropriate for the TRA perched-water system. It is not expected that alternate sources of recharge have a large effect on simulated contaminant concentrations or it would have been difficult to calibrate the model to observed concentrations without accounting for those recharge sources.

Two assumptions relating to the period of CWP discharge and the rate of discharge are invalid based on current knowledge of TRA facility operations. The model assumed that discharge to the CWP would continue through 2007. It is now known that the TRA facility will operate at least 20 years longer, and CWP discharge at a rate of approximately 300 million gal/yr will continue during that interval.

7.5.3 Update of the Pre-Record of Decision Perched-Water System Model

The TRA perched-water system model was updated to include the discharge of wastewater to the CWP through 2024. In addition, radioactive decay half-lives for Co-60 and Sr-90 were implemented in each case, because they were apparently omitted during simulation with the original perched-water system model. Results from the updated model show the maximum concentrations of all COCs in the SRPA are expected to be well below MCLs by 2095. The Co-60 and Sr-90 concentrations are predicted to never exceed MCLs. Concentrations of the contaminants in the SRPA are predicted to fall and remain below MCLs by 2115—the date for which risk to future site users is defined in the OU 2-12 ROD (DOE-ID 1992)—and continued discharge of water to the CWP is expected to be protective of human health and the environment under the conditions presented in the ROD (i.e., groundwater monitoring with comparison to predicted concentration trends) and assumptions employed in the original, updated, and extended perched-water system models.

8. SUMMARY

The First Five-Year Review Report (DOE-ID 2003) concluded that implemented remedies from the OU 2-13 ROD (DOE-ID 1997a) are protective of human health and the environment. The review observed that COCs measured in the aquifer during this first five-year review period either are currently below the MCLs or are projected to be below the MCLs in 2012. The First Five-Year Review Report also identified a set of issues and recommendations. This document details actions taken in response to those issues and recommendations.

The First Five-Year Review Report (DOE-ID 2003) identified the sporadic recurrence of free-phase diesel fuel in the PW-13 TRA perched-water well as an issue. Two new downgradient wells (TRA-1933 and TRA-1934) were installed to investigate the extent of the diesel contamination. Samples also were collected from selected existing perched-water and aquifer wells to test for the presence of dissolved-phase diesel components. Diesel components were not detected in the selected aquifer wells. Relatively low concentrations of dissolved-phase diesel were measured in several of the perched-water wells. Small amounts of free-phase diesel were detected in the two new perched-water wells and in PW-13. The diesel fuel appears to have aged in contact with the perched-water body, and no new sources of diesel fuel were identified. The free-phase diesel fuel in the vicinity of PW-13 is apparently from a 1981 spill of 2,000 gal, as previously concluded in a Track 1 investigation of the responsible fuel transfer line. The sporadic recurrence of free-phase diesel fuel in Well PW-13 is most likely the result of a natural cycling or remobilization mechanism in the fractured basalt/sediment sequence that is driven by changing water levels. The remaining diesel fuel likely will continue to dissolve and disperse in the perched-water body. The continued presence of the perched-water body helps to isolate the diesel fuel from the aquifer. The rate of natural attenuation is expected to slow as the more soluble components of the free-phase diesel are preferentially removed, leaving behind the less easily degradable fraction.

The First Five-Year Review Report (DOE-ID 2003) recommended that a geochemical investigation be performed to "fingerprint" various water sources at TRA and correlate those sources to water samples collected from the perched-water wells. Data were collected from selected wells in both the perched water and the aquifer. The contaminant, water-quality, stable-isotope, and water-level data indicate that the deep perched water consists of multiple zones rather than a single, continuous deep perched-water body. Some of the perched-water wells within TRA appear to have a water source other than the CWP. Wells PW-12, PW-13, and USGS-072 appear to reflect leaking (raw water) piping in combination with some precipitation input. Well USGS-068 reflects contamination from the former Chemical Waste Pond in combination with a local water source (precipitation and raw water lines). Although the CWP is not the source of contaminants, water infiltrating from the CWP is the primary source of water for perched water beneath TRA. Infiltration from the CWP appears to aid in the migration of contaminants through the vadose zone to the aquifer since the wells in the SRPA that have the highest contaminant concentrations (tritium and chromium) also show the strongest influence from the CWP.

The First Five-Year Review Report (DOE-ID 2003) identified the unexplained increase of Co-60 in the PW-12 perched-water well that occurred between October 2001 and March 2003 as an issue. More recent samples from October 29, 2003, and March 17, 2004, show a continuing decline in the Co-60 concentration at Well PW-12. Potential sources were examined. A review of the area surrounding PW-12 indicated a history of contamination and presence of Co-60. Three OU 2-13 CERCLA sites are located within 100 ft of PW-12. The three sites were all known or believed to have Co-60 contamination present in varying levels. Several abandoned warm and hot waste pipelines, including two with a history of releases (responsible for the three CERCLA sites), are located within 100 ft of PW-12. The increase in Co-60 likely is the result of a pulse of residual contamination to PW-12 or the result of changing subsurface conditions. The contamination is probably transported down to the perched zone by water from a leaking raw water line with precipitation possibly aiding the transport of contaminants to PW-12.

The geochemical signature of the perched water at PW-12 supports the belief that a leaking utility line transports the contamination.

The First Five-Year Review Report (DOE-ID 2003) identified the unexplained steady or increasing activities of Sr-90 in the PW-12, USGS-054, USGS-055, and USGS-070 perched-water wells as an issue. It was further recommended that the following mechanisms be evaluated as being potential causes for these unexplained trends: (1) adsorption/desorption occurring with changing perched-water levels, (2) changing flow pathways in response to remediation and fluctuations in discharge to the CWP (or alternating cells), (3) seasonal variations of natural infiltration at a local scale, (4) variations in recharge from unidentified manmade sources, (5) lateral flux from the Big Lost River, or (6) new leaks of contamination from unidentified sources. In addition to these six mechanisms, recent research suggests that the fundamental physics of unsaturated flow in fracture networks can lead to pathway switching and temporal fluctuations in downwards flow. Alone, or in combination, the aforementioned mechanisms will induce localized temporal fluctuations in measured concentrations within a generally declining trend. Given the complexity of the subsurface environment and spatial/temporal variability in both the amount and chemistry of recharge sources, short-term variations in concentration levels are to be expected for the perched-water body. These short-term variations are not of concern as long as the trend is seen to decrease again over a maximum period of a few years. At the time of the five-year review, it was also believed that the TRA-605 Warm Waste Line, known to have released contamination, was contributing to the increasing Sr-90. A Track 2 investigation into the extent and nature of the contamination is currently being conducted. Findings from this investigation will be presented in a Waste Area Group 10 Track 2 Summary Report in Fiscal Year 2005.

The First Five-Year Review Report (DOE-ID 2003) identified continued usage of the CWP beyond 2007 as an issue. At the time of the OU 2-13 ROD (DOE-ID 1997a), it was assumed that the TRA, including the CWP, would be decommissioned in 2007. Under a recent decision (2003) by DOE, the TRA will remain active for at least another 20 years. The TRA perched-water system model was updated to include the discharge of wastewater to the CWP through 2024. In addition, radioactive decay half-lives for Co-60 and Sr-90 were implemented, because they were apparently omitted during simulation with the original perched-water system model. Results from the updated pre-ROD model for the 20-year case show that the maximum concentrations of all COCs in the SRPA are expected to be well below MCLs by 2035.

The Co-60 and Sr-90 concentrations are predicted to never exceed MCLs. Because concentrations of the contaminants in the SRPA are predicted to fall and remain below MCLs by 2115, the date for which risk to future site users is defined in the OU 2-12 ROD (DOE-ID 1992), continued discharge of water to the CWP is expected to be protective of human health and the environment under the conditions presented in the ROD (i.e., groundwater monitoring with comparison to predicted concentration trends) and assumptions employed in the original, updated, and extended perched-water system models.

9. RECOMMENDATIONS

This report recommends the following actions to ensure that the selected remedies are protective of human health and the environment and to ensure best management practices:

- Installation of petroleum traps in Wells PW-13, TRA-1933, and TRA-1934. The traps will allow the volume of floating diesel to be measured and provide an inexpensive, passive means for the removal of old diesel and collection of samples for analyses to determine if new diesel is entering the subsurface. The petroleum traps should be maintained monthly with concurrent measurements of water level and diesel thickness. A sample of the floating product should be collected (if sufficient quantity reappears) and analyzed to verify that the less than C_8 fraction of the product is gone and that it is composed primarily of greater than C_8 hydrocarbons.

- Periodic monitoring of selected perched-water wells and aquifer wells for dissolved components of diesel fuel may be warranted, but it is recommended that such measurements be made at least annually, unless monitoring results dictate otherwise.

- Continued monitoring of Well PW-12 for Co-60 in accordance with the approved Groundwater Monitoring Plan for TRA.

- Update the Groundwater Monitoring Plan (DOE-ID 2004a) to include the two new wells installed as part of this investigation, TRA-1933 and TRA-1934, and to add diesel-range organics to the analyte list for PW-12 and to add USGS-073 to the monitoring plan for diesel-range organics analysis.

- Monitor VCO investigations of the piping systems at TRA in relation to Co-60 activities at Well PW-12 and to aid in developing long-term understanding of the perched-water system beneath TRA.

- Correlate the stratigraphic and lithologic structure of the TRA subsurface with recent geochemical fingerprinting that indicates multiple and distinct sources for the perched water. Previous interpretations of the TRA perched-water system referred to only two distinct water bodies: the shallow and the deep perched. The significance of multiple deep perched-water bodies is unclear at this time. Developing an enhanced understanding of the perched-water bodies might provide additional insight into their influence on contaminant transport.

- Continued monitoring of the perched water wells according to the existing groundwater monitoring plan with modifications as approved by DOE, DEQ, and the EPA. It should be realized that not all increasing trends or spikes in contamination pose an immediate or eventual threat to the effectiveness of the remedy and should be evaluated individually to determine the potential of their impact.

- Keep abreast of and support ongoing research (e.g. Vadose Zone Research Park and old INTEC percolation ponds) that focuses on further development of improved conceptual models of unsaturated flow in fractured rock environments and research that focuses on the development of numerical simulators that can take advantage of new advances in these conceptual models.

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